Fishing for Litter: A cost-benefit analysis of how to abate ocean plastics.

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Fishing for litter: A cost-benefit analysis of how to abate ocean pollution

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ES50064 MRes Dissertation

Sub Questions:

- *Is it financially viable to recover marine debris?*
- *Is it economically viable to recover marine debris?*
- *Can adjusted unit value transfer be used to estimate the damages of marine debris?*

Abstract

Marine plastics inflict annual damages of nearly $2bn on the marine economy. These damages may now be mitigated by a novel scheme that proposes to recover marine plastics. While a range of uses exist for recovered marine plastics; this research finds that none are profitable. While not financially viable, the scheme is found to be economically viable given the significant damages from marine plastics on the marine economy. Implications from this research for the future of pollution abatement are remarked upon.

Keywords: Cost-benefit analysis, recycling, marine plastics.
JEL classification: D26, H23, Q58

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Introduction

The 620% rise in plastic production since 1975 has been accompanied by an increased volume of plastics in the world's oceans (Jambeck et al., 2005). Indeed, of the estimated 6.4 million tonnes of marine debris that enter the oceans annually, between 60 and 80% is believed to be plastic (Derraik, 2002). Once in the ocean, marine plastics inflict adverse effects on both marine life, via ingestion, entanglement and leached chemicals (Azzarello and Van-Vleet, 1987), and the marine economy, via damaged vessels, lost fishing and reduced tourism (Jambeck et al., 2005). While there is some ambiguity about how fatal plastics are to marine life, there is considerable support for the view that marine plastic inflicts significant damages on a range of sectors in the marine economy (McIlgorm, Campbell and Rule, 2011). Therefore, to abate the damages caused by the growth in plastic production, some method of recovering marine plastics must be sought.

To recover marine plastics, a variety of methods have been proposed. The most notable of these was devised by a new start-up, called 'The Ocean Cleanup' ('TOC'), that developed a novel technology that purports to be able to recover floating marine plastics. The technology essentially uses a boom, deployed across the North Pacific, to force marine plastics to converge and accumulate in the area of the boom. The boom may then recover floating plastics and store them, ready for transport to shore. Once ashore, recovered marine plastics may be used in a manner of ways, such as generating oil (pyrolysis), electricity (waste-to-energy) or new plastics (recycling) and TOC believes that these uses may be profitable. However, this research fails to support this assertion and instead notes that the low market prices of oil, electricity and plastics cannot support the high operating costs of this scheme.

While the operating costs exceed the revenues, the economic benefits also exceed the costs. This suggests a duality in the viability of TOC whereby the scheme is shown to be economically but not financially viable. This result was found by conducting a cost-benefit analysis (CBA) of the scheme. CBA was used first to determine financial viability and then extended to explore economic viability. To determine financial viability, the expenses were updated from a prior feasibility study on TOC and then compared to revenues estimated using market prices. To determine economic viability, these costs and benefits were augmented by a calculation of the avoided damages from marine debris. To estimate marine debris damages, previous estimates of marine debris damages were subject to adjusted unit value-transfer. This technique has been commonly used to ascertain valuations where primary data is not available or previous estimates are inexact (Brouwer and Spaninks, 1999). Additionally, once adjusted, new values were discounted and subject to sensitivity analysis to increase the accuracy and validity of the estimates. The estimates produced can be summarised by a ratio of financial costs to revenues of 1:24, implying the scheme is not financially viable, and a ratio of economic costs to benefits of 1:2, implying the scheme is economically viable. The implications of this duality are explored with regards to suitable funding models.

This research is organised as follows. Firstly, a review of the literature suggests uncertainty about the effects of marine debris and how to possibly mitigate them and the TOC scheme was selected to mitigate them. To examine TOC's financial and economic viability, an appropriate method was designed given the restrictions of no primary data and scarcity of secondary data. This method was implemented in the results section which details the financial and economic costs and benefits of the scheme. Finally, comment was made on the limitations and implications of this research.

Literature Review

This section explores the literature on the effects, control, types and uses of marine debris. Marine debris may enter the oceans either as industrial effluent (Roberts and Spence, 1976), mismanaged consumer waste (Azzarello and Van-Vleet, 1987), or from dumping at sea (Hammer, Kraak and Parsons, 2012). Once in the ocean, there are three main effects of marine debris on marine life; ingestion, entanglement and leaching. Additionally, there are three main effects of marine debris on the marine economy; damage to vessels, lost fish stocks, and reduced tourism (McIlgorm, Campbell and Rule (2011), Mouat, Lozano and Bateson (2010)). However, there is considerable uncertainty in the literature regarding the magnitude of these effects. This section, therefore, reviews the literature in this area. This review is then expanded to review the literature in the related areas of how to control and use marine debris.

Effects of marine debris

Ingestion.

This section discusses the effects of marine debris on marine life. The most well-known danger is that of ingestion. For marine life that mistakes marine plastics for their typical diet, there are two significant dangers of ingestion. Firstly, large items may occlude both respiratory and digestive tracts leading to starvation or asphyxiation. However, the likelihood of this danger is low as a majority of marine plastics are small industrial pellets or fragment at sea and so cannot cause such harm at the reduced size (Karlsson et al. (2018), Robards, Piatt and Wohl (1995)). The second danger is, however, more fatal. Once ingested, plastics are digested exceptionally slowly which can leach toxic chemicals into the body. However, this effect is uncertain as there is insufficient evidence thus far to corroborate this suggestion (Azzarello and Van-Vleet, 1987). Additionally, decade-long surveys by Moser and Lee (1992) and Robards, Piatt and Wohl (1995) both failed to report any adverse health issues from ingestion. While these two studies only observed seabirds, a wide-ranging survey undertaken by Azzarello and Van-Vleet (1987) reported the following conclusion:

"Although many of these potential hazards lie in the realm of conjecture, we believe that it is important to work toward prevention or prediction of these problems before the actual effects have been realised." (Azzarello and Van-Vleet, 1987, p. 301)

While the fatality of ingestion is ambiguous, the frequency and volume of plastics ingested is more certain. Moser and Lee (1992) produced strong evidence for this as they reported on the same population of seabirds in 1969-77 and 1978-85, and found that the frequency of ingestion had risen by 30% and the volume of plastics ingested had risen 80%. This evidence was further corroborated by Robards, Piatt and Wohl (1995) who also reported a statistically significant increase in both frequency and volume of ingested plastics between 1968-77. Additionally, Robards, Piatt and Wohl (1995) noted a significant difference in the likelihood of seabirds ingesting marine plastics in the Atlantic and the Pacific which implies that the increasing danger of ingesting marine plastics has grown inconsistently across the worlds oceans. Robards, Piatt and Wohl (1995) attributed this increasing but inconsistent danger of ingestion to an increase in the frequency and volume of marine plastics in the ocean. Jambeck et al. (2005) noted this growth as being consistent with the trends in global plastic production. Therefore, the increasing rate of ingestion is clearly linked to the increasing concentration of marine plastics (Lebreton et al., 2018). This implies support for a scheme that would reduce the concentration of marine plastics and therefore reverse the current increasing trends of ingestion.

Entanglement.

Another danger from marine debris is that of entanglement. Once entangled, marine life may suffer either strangulation or starvation. Unlike the dangers from ingestion, these effects have been commonly observed as the rates of species suffering from entanglement (40%) is notably higher than those suffering from ingestion (30%) (Allsopp et al., 2007). While it is unlikely that a majority of marine plastics will entangle marine life due to fragmented size, there are more types of marine debris than plastic. Specifically, discarded fishing gear, or ghost nets, are a common type of marine debris that are responsible for a majority of marine life suffering entanglement. Ghost nets were initially designed to entangle marine life and continue to do so when discarded (Henderson, 2001). They are discarded either accidentally or dumped after losing economic value. This loss of economic value suggests that recovering ghost nets cannot be profitable. Therefore it is unlikely that ghost nets would be recovered by the private sector which would have reduced entanglement. However, as previously noted, there is a significant loss to marine life caused by entanglement which suggests recovery of them is warranted. This dichotomy between damages and viability is explored in greater depth later.

Leaching.

A final notable danger from marine debris on marine life is that of leaching chemicals. Once exposed to a range of factors including weathering, oxidation and UV radiation, marine plastics have been observed to leach chemicals. The chemicals leached may be part of their design, such as dyes or fire-retardants, or may be adsorbed from the water. In highly concentrated areas of marine debris, chemicals now banned for their toxicity, such as PCB and DDT (Harse, 2011), may still be present and may be adsorbed by new marine debris in the area. Once adsorbed and then leached, the dangers of reduced water quality and ingesting marine plastics are exacerbated. Once ingested, the process of digestion implies plastics beginning to leach contaminants which are likely to induce significant health risks. These health risks may be transmitted through the food chain. However, with only 30% of species ingesting plastics, this effect is likely to be limited (Allsopp et al., 2007) and even then there is little supporting evidence for the existence of this effect (Azzarello and Van-Vleet, 1987). Additionally, the dangers from reduced water quality may still abound although significant health effects of reduced water quality due to marine pollution have not yet been reported. However, this is tempered by the observation from Harse (2011) that:

> **"No fish-monger on earth could sell you a certified-organic wild-caught fish" (Harse, 2011, p. 341)**

Therefore, the leaching of plastics in the ocean is a threat to marine life. However, it is potentially possible to mitigate this threat. It should be noted that while only 15% of marine debris floats, 80% of that is plastic of which 90% is the necessary type to float in water (Sudhakar et al. (2007), Derraik (2002)). Therefore it is fair to suggest that a majority of plastics float. Floating plastics are more easily recovered by mitigation schemes such as TOC. However, as they float, they are also more exposed to the range of conditions that degrade them significantly enough to start leaching chemicals. Therefore the scheme is most likely to recover those plastics that are most at risk of or have already begun, leaching chemicals. Furthermore, Sudhakar et al. (2007) noted that plastics might retain a degree of economic value even when significantly degraded and therefore it is possible that it may be financially as well as economically viable to recover floating plastics (Hammer, Kraak and Parsons, 2012). The implications of this finding from the literature are discussed more fully later, but for now, a discussion of the dangers of marine debris on the marine economy is merited.

Damages to economy.

The discussion above has made it eminently clear that there is a range of threats from marine debris to marine life. These effects also inflict damages on the marine economy. Specifically, the problems of ingestion, entanglement and leaching effectively reduce fish stocks which therefore reduce the catch and revenue of the fishing economy (Azzarello and Van-Vleet, 1987). These losses are so severe that Mouat, Lozano and Bateson (2010) reported that the Scottish fishing fleet lost an estimated 5% of their total annual revenue to marine debris. While the Scottish fishing fleet faces marine debris in the Atlantic Ocean, and the North Sea so is more exposed to these effects, their experience is not atypical with Takehama (1990) reporting damages to the Japanese fishing fleet also in the millions of dollars. Whether this was comparable to the 5% of annual revenue figure is unclear, but the finding of significant damages from marine debris is consistent.

Further dangers to the marine economy were evident from Mouat, Lozano and Bateson (2010) and McIlgorm, Campbell and Rule (2011) who both reported that a range of sectors suffered from marine debris including fishing, as discussed above, but also vessels and tourism. Specifically, the most frequent damage suffered was that on equipment and vessels while the most significant damage was the estimated losses suffered by the tourism industry. Lost tourism revenues represent a significant cost to coastal populations despite their limited exposure to marine debris as only 15% of marine debris washing ashore as beach debris (Derraik, 2002). Once ashore, beach debris reduces aesthetic value of an area imposing a cost in lost revenue. The cost was so significant that just a 25% reduction in beach debris was predicted to yield approximately $30mn in benefits to the local area. While the magnitude of this effect may be disputed as their results were based on the very popular Orange County USA area, the finding that marine debris reduces tourism is a common one in the literature (Mouat, Lozano and Bateson, 2010). Therefore it is clear that marine debris imposes significant costs on a range of sectors in the marine economy. Specifically, losses to fishing and tourism highlight the significant economic impacts of marine debris.

The discussion above has outlined the various adverse effects of marine debris on both marine life and economy. This implies that it is necessary to recover marine debris to mitigate these damages. The following discussion reviews the literature on plausible methods of mitigating marine debris.

Control of marine debris

Legislation.

Efforts to legislate marine debris have thus far been ineffective in part due to the challenges of the international nature of marine debris and the link to economic growth. As economic growth is accompanied by increased production and waste, there is an active link between economic growth and marine debris which dissuades developing countries from regulation who are loathed to sacrifice economic growth in favour of environmental protection (Jambeck et al., 2005). Furthermore, the challenges of international cooperation and enforcement have hampered previous efforts. The effect of this was that current legislation has ineffectively focused on the input rather than the stock of marine debris (Amos, 1993). An illustrative example is MARPOL Annex V, the International Convention for the Prevention of Pollution from Ships, which came into force in 1989 and regulated the dumping of waste from ships at sea. However, the effect of this regulation has not been uniformly acclaimed in the literature. Henderson (2001) found no statistically significant difference in entanglement rates from 1987-1996 despite the legislation beginning in 1989. Henderson (2001) studied Hawaiian monk seals which reside on Hawaiian Islands which are proximate to where TOC will deploy. However, two limitations to their study exist. Firstly, they did not observe any fish, whales, or birds in their study as other entanglement studies have done (Laist, 1997). Secondly, they did not cover entanglement rates at sea. This is crucial as the frequency of illegal dumping is higher at sea, and so their sample had a priori limited exposure to entanglements. However, this was a purposive sample selection as the use of an island based sample is instructive both for quantifying the risks to land-based life from marine debris and also as the justification for the clean up of beaches. However, their finding of no significant difference in entanglement rates pre and post-legislation is evidence of some success. As entanglement rates have risen sharply, commensurate with the growth in plastic production and marine debris (Jambeck et al., 2005), finding no increased rate of entanglement could suggest that the legislation had some small effect as entanglement rates should have risen if there was indeed no effect (Laist, 1997). Therefore there is insufficient evidence to conclude that this legislation had any significant effect on illegal dumping of marine debris and resultant entanglement rates and losses of marine life.

Beaches.

Beach cleanups, volunteer efforts to recover beach debris, have thus far proved to be the most popular method of recovering marine debris (The Ocean Cleanup, 2014). Beach debris reduces the aesthetic value of an area, reducing tourism, and reduces the quality of the beach, reducing health (Leggett et al., 2014). The effect on tourism is significant despite beach debris representing only 15% of total (Derraik, 2002). Therefore, controlling beach debris can be characterised as an investment in the local economy (Leggett et al., 2014). The return on this investment is significant with Leggett et al. (2014) estimating that a 25% reduction in beach debris in Orange County USA would yield a $32.2 million benefit to the local economy. However, the cost of achieving such a reduction is uncertain. With volunteers, the recovery cost per kilogram of waste recovered can be incredibly low which suggests that it may be profitable to run such schemes. Profitability here refers to both the sale of recovered plastics and in increased or maintained tourist revenues. This suggests an essential difference between financial and economic viability which is explored further later. However, such cleanups are also slow, laborious and require a significant amount of manpower to make the clean up worthwhile. Therefore, it is unlikely that a labour force sufficient to cover the sheer scale of coastline affected may be consistently found unless they are paid which increases the costs of running the operation. Paid efforts are familiar with local authorities in the UK reporting the expenditure of more than $35,000 annually to keep areas clear of beach debris (Mouat, Lozano and Bateson, 2010). However, the efficiency of this method is debatable. Specifically, Leggett et al. (2014) noted that within three months, there had been a 50% return to the original level of beach debris which implies repeated expense while the source of debris is unaffected. Furthermore, beach debris represents only 15% of the total stock thus suggesting a limited impact from beach cleanups. Finally, this method abates damages to tourism more strongly than any other sector, despite other sectors, such as fishing and shipping, also significantly being affected by marine debris. This suggests that it is not a practical solution to the more significant problem of marine debris.

Drones.

To mitigate the damages of marine debris via recovery of marine plastics at sea, several proposals have been made. One of the most innovative of these suggests using autonomous drones. The drone concept proposes using several small vessels remotely or automatically salvaging marine plastics and then returning to a set drop-off location afterwards. The main advantage of this scheme is flexibility in targeting recovery efforts (The Ocean Cleanup, 2014). Possession of a small fleet of drones allows more heavily concentrated areas or areas of higher risk to be specifically targeted. The possibility of responding to localised events, such as the millions of tonnes of debris washed out by the tsunami in Japan (Jambeck et al., 2005), also arises. Another considerable advantage to the drone concept is the notion of autonomy. As drones may be automatic and solar panel fuelled, they require a low degree of human interaction, reducing operating costs. However, several drawbacks persist. Firstly, design and repairs to such a fleet could prove both challenging and costly if the drones frequently break or overload in the notoriously challenging conditions in the ocean such as strong currents and highly concentrated areas of marine debris. Furthermore, each drone would have to recover a certain amount of plastic per operation for it to be profitable and therefore a tension arises between the operational limits of remote control, the battery capacity, and the profitable level of operation. These criticisms are also compounded by considerable uncertainty regarding the questions of how large should each drone be, how should each drone capture only marine plastics and where and how would each drone return to after cleaning an area. So despite the futuristic nature of this suggestion, it is fraught with challenges (The Ocean Cleanup, 2014).

Vessels.

Discarding the drone concept as insufficiently specified leaves two further noteworthy options. Either large manned vessels or use of passive systems. The former has been labelled as fishing for litter (KIMO, 2014). Operated in the North Sea by KIMO, this scheme pays fisherman for marine debris they recover as part of their regular fishing activities (KIMO, 2014). Indeed, with 86% of fishing vessels reporting marine debris contaminated bycatch (Mouat, Lozano and Bateson, 2010), this is likely to be a popular scheme in the fishing industry. Indeed the preliminary trial evidence is also supportive of the scheme, although this was found only on a small-scale trial which had governmental support (KIMO, 2014). Governmental support for this scheme is warranted by the fact that it has shown, thus far, that it is effective at recovering all kinds of marine debris, including the low financial but high economic value ghost nets. These may not always be plastic or have resale value, and so revenues from them are likely to be inefficiently low. Additionally, with average costs of just under \$2.54 per kilo of waste collected (The Ocean Cleanup (2014), KIMO (2014)), inefficiently low revenues for the scheme predicate the scheme being financially infeasible. However, the high damages suffered from marine life becoming entangled in ghost nets suggests that recovering them may be at least economically viable if not financially (Azzarello and Van-Vleet, 1987). To adapt to this duality, there is some suggestion that governments could support the scheme by hypothecating some new revenue stream towards funding such a scheme. A new revenue stream could hypothetically be from market-based instruments for control of marine plastics which would be an example of environmentally proactive policy (Dikgang, 2012). While this is theoretically possible, a more likely degree of government support would be aiding the scheme in managing recovered marine debris. Specifically, existing waste management infrastructure, particularly waste-to-energy plants (DEFRA, 2014), could theoretically accept marine debris as well as municipal waste. This would provide a specific, consistent use for recovered marine debris.

It is, however, uncertain whether this scheme is viable. Specifically, three severe disadvantages persist for the scheme. Firstly, recovery of plastics is much more challenging for regular fishing activities given that a majority of plastics, 70% (Derraik, 2002), sink and are therefore irrecoverable (Derraik, 2002). Secondly, fish has an established market while recovered marine debris does not which is a disincentive to fishing for them. Even among the uses for recovered plastics, there is also a high degree of uncertainty regarding their market value due to the effect of degradation at sea reducing their suitability for a range of economic uses (Sudhakar et al., 2007). Thirdly, calculations by KIMO (2014) reveal that the total catch of all the fishing for litter trials is too low to have any significant impact on the amounts of marine debris in the ocean. Therefore, the fishing for litter scheme does have some merit as being economically viable although it is too small a scale and too costly to be financially viable. The small-scale relegates its applicability to the broader problem of marine debris. This finding leaves one viable option for clean up, the design of a passive system which may passively recover marine plastics.

The Ocean Cleanup.

The TOC scheme is one such passive system which is the focus of this research. The TOC project suggests the construction, essentially, of a large-scale boom which passively sits in an area of highly concentrated marine debris. Once there, floating plastics are forced to converge to a single point where they are then recovered and stored. Infrequently, a manned ship would recover the stored debris and transport it to land for marketing. Predictably, the main challenges to this scheme are in the technical design of such a boom. Assuming this can be surmounted, indeed reports consequent to the feasibility study suggest that this is a valid assumption (The Ocean Cleanup, 2014), then the scheme's central costs would be the transportation of recovered waste from the boom to a land-based site. This suggests that TOC could operate at much lower cost and much larger scale than the previously discussed alternatives.

However, there are some notable disadvantages. Firstly, TOC proposed only recovering floating plastics which restricts the recovery efforts to the 15% of marine debris that floats (Derraik, 2002). While this ignores the 70% of marine debris that sinks, it is likely a profitable move as the higher value plastics are those that typically float (The Ocean Cleanup, 2014). Therefore, a drawback to the scheme is that it will have a limited effect on the 70% of marine debris that sinks which implies that the scheme will have a limited effect in mitigating the economic impacts of marine debris (Derraik, 2002). However, this is addressed by stating that floating plastics are those most likely to be ingested as they are easier to see for marine life when at the surface, and also most likely to leach chemicals as there are stronger degradation factors on the surface (Azzarello and Van-Vleet (1987), Harse (2011)). This suggests that while only 15% of debris may be recovered, this represents the most dangerous proportion of marine debris. It should also be noted that 15% of marine debris that may be recovered by TOC may not necessarily be plastics. This implies that some method of sorting is required as recovered plastics must be examined for their suitability for the market. Specifically, pyrolysis and recycling require low levels of degradation while waste-to-energy and landfill have much lower requirements (Sudhakar et al., 2007). Furthermore, contaminants and by-catch must further be eradicated from recovered marine debris to ensure maximum yield and profit from the recovery of marine debris. Such checks on recovered marine debris increase the costs of the scheme. Given these costs and the uncertain costs of the necessary research and design for the scheme, the cost of recovery is likely to be significant using TOC which casts doubt on the profitability of recovering marine plastics.

Types of marine debris

With regards to the profitability of recovering marine plastics, it is instructive to explore the types of marine debris that might be recovered. Within the 60-80% of marine debris that is estimated to be plastic (Derraik, 2002), three categories of plastics emerge; floating and recyclable, floating and unrecyclable, and sinking. While only 15% of marine debris floats (Robards, Piatt and Wohl, 1995), this category is comprised of the most commonly produced plastics, Polyethylene ('PE'), polypropylene ('PP') and polystyrene ('PS'). With regards to the frequency of these plastics, sample evidence suggests that 90% of marine plastics are polyethylene ('PE'), distantly followed by polypropylene ('PP') (Karlsson et al., 2018). The source of these plastics are both industrial, with Karlsson et al. (2018) noting that billions of PE pellets might be found in industrial effluent, and commercial, with Kothari, Tyagi and Pathak (2010), noting that all three types are commonly used for goods packaging. Regardless of the source, once these plastics enter the ocean, they float and thus pose a threat to seabirds and fish via ingestion. However, as they float, this threat may be more easily mitigated as floating plastics are more accessible to observe and recover. Recovery of these plastics is essential as PE, PP and PS are all suitable for recycling alongside pyrolysis and waste-to-energy uses which suggests a degree of profitability to their recovery (Psomopoulos, Bourka and Themelis, 2009). This further implies that management of these may be integrated easily into existing waste management infrastructure. This is discussed in greater detail later, but for now, it suffices to note that these plastics are the most common, the most valuable, and the most recoverable.

The second category of marine plastics is predominantly polyvinyl chloride ('PVC', often used in construction). PVC floats in water similarly to PE although has lower financial value. Specifically, The Ocean Cleanup (2014) reports that there are limitations to PVC's viability for pyrolysis and the low number of times it may be recycled. This implies that it has a lower financial value than the first group. This lower financial value is compounded by the need to ascertain the value of recovered PVC which would require some form of screening which adds to the cost of recovery (The Ocean Cleanup, 2014). This higher cost and lower value compared to the first group implies reduced financial viability to its recovery. However, this distinction is not practical as with both PE and PVC floating it is not feasible to recover only the valuable PE at the expense of the dubiously valued PVC. Additionally, as PVC has been observed to leach chemicals more quickly than other plastics, there is added impetus to its recovery (Sudhakar et al., 2007). The degradation implies reduced financial value to its recovery although the increased damages threatened by degraded plastics imply an increased economic necessity of its recovery. Indeed this dichotomy between financial and economic viability is a critical theme in this research. To summarise, PVC is less financially viable to recover but is more economically viable (The Ocean Cleanup, 2014).

Finally, a third group of marine plastics exist which mainly includes Polyurethane ('PU'). PU sinks in water and cannot be used for either pyrolysis or recycling. The lack of commercial use for PU implies little to no financial value. This low value has the effect of disincentivising the necessary developments required for the scheme to be able to recover sunk plastics given the depth of the Pacific (The Ocean Cleanup, 2014). Finally, it should be noted that while sunk plastics are less exposed to degradation, they are more likely to inflict damages on lower level marine life, so it is at least, somewhat economically sensible to recover such plastics. In conclusion, sunk plastics, while rare, are unlikely to be recovered due to low cost and technical limitations. This concludes the discussion of marine plastic types although it is now necessary to discuss the proportion of marine debris which comes not from plastics but ghost nets.

A sizeable proportion of marine debris is not, however, plastic as detailed above, but discarded fishing gear, referred to previously as ghost nets (Hammer, Kraak and Parsons, 2012). The discussion on legislation of marine debris has already discussed how ghost nets are dumped at sea due to low economic value

(Henderson, 2001). Once they have lost value, fishing vessels, in particular, may dump or lose ghost nets, which represents an abundant source of marine debris (Marsden Jacob, 2009a). Indeed the volume of ghost nets in the ocean has increased recently with a growth in the fishing industry (Amos (1993), Johnson (1994)). The impacts of ghost nets on the marine economy are to entangle all manner of marine life which devastates fish stocks. Fish in highly concentrated areas of marine debris are effectively being fished by ghost nets without any fishing activity occurring. These dangers are exacerbated by ghost nets transporting invasive or dangerous species across layers of the water column (Derraik, 2002). However, this has not yet been observed on a large scale (Azzarello and Van-Vleet, 1987). Finally, these dangers are compounded by the estimated 50-year lifespan of ghost nets and the lack of incentive to recover these nets. This is because as the material degrades the nets are not worth keeping for vessels which then dump them at sea. Therefore it is unlikely that any financial value may be realised from recovering ghost nets, implying that cleanup efforts are unlikely to address this issue (Marsden Jacob (2009a), Hammer, Kraak and Parsons (2012), Henderson (2001), Azzarello and Van-Vleet (1987)).

This review has so far explored a range of impacts from marine debris, a range of mitigating options and the range of debris to be found. What remains to be discussed is the range of uses for recovered marine debris once recovered.

Uses for marine debris

Pyrolysis.

The first main use for marine debris is pyrolysis, the process of turning plastics into oils. Pyrolysis transforms recovered plastics into crude oil for use, sale or refinement. The decision to use, sell or refine the oil is dependent on the comparison of refinement costs to oil prices. Oil prices are notoriously volatile however which introduces a significant degree of uncertainty for this. Further uncertainty arises about the viability of using pyrolysis for marine debris as not all types and quality of marine debris are suitable for pyrolysis. While the most common PE, PP and PS types are usually acceptable, this is provided that they not be significantly degraded, else they must be discarded introducing a loss (Sudhakar et al., 2007). Furthermore, the more infrequently recovered PVC is not typically accepted for pyrolysis nor is non-plastic marine debris such as ghost nets, which physically cannot be used for pyrolysis but are responsible for the majority of losses of marine life to entanglement, (The Ocean Cleanup, 2014). This illustrates that pyrolysis is only a valid option for plastics and not other types of marine debris which would else have to be landfilled and therefore this is a less efficient use for recovered debris. Furthermore, these restrictions reduce the effective yield from recovery which reduces estimated profits (Fivga and Dimitriou, 2018). Another factor that reduces profits is a scarcity of such plants. As pyrolysis can only be used for plastics, there are few pyrolysis plants available, and therefore there is limited scope for integrating marine debris management with existing waste management infrastructure (Psomopoulos, Bourka and Themelis, 2009). This scarcity of plants may also increase the costs of recovery if debris requires transportation to more efficient plants and more profitable markets. To conclude, there is a significant degree of uncertainty regarding the viability of using pyrolysis for recovered marine debris.

Waste-To-Energy.

A similar but less restrictive option to pyrolysis is called Waste-To-Energy (WTE) which proposes that recovered plastics be incinerated to generate electricity for use and sale. WTE is preferred to pyrolysis as a higher amount of recovered marine debris could be used in a WTE plant than a comparable pyrolysis plant. This is because the pyrolysis equivalent may only use sufficiently non-degraded plastics whereas WTE may accept nearly all types of waste (DEFRA, 2014). Furthermore, as WTE plants are currently integrated into the waste management infrastructure, the addition of marine debris to these plants would appear to be of negligible difficulty. Such integration is favoured by decision-makers who are spared the expense of constructing new plants. Existing WTE plants also confer power generation from waste with is further attractive to policymakers. Precisely, once marine debris management is assimilated with existing infrastructure, it represents a greener source of local power generation once integrated into the existing system. While this is an attractive option as there are fewer restrictions on what debris is accepted and marine debris may be easily integrated into existing infrastructure, several issues with its viability arise. Firstly, transport costs would be higher than other options as very few WTE plants currently exist (Kothari, Tyagi and Pathak, 2010), especially when compared to the more common recycling plants (DEFRA, 2014). Secondly, the volatile price of electricity is a significant concern for the viability of using WTE. As the electricity price is strongly related to a range of factors including spare capacity on the grid, and the localised demand at different times of the day (Psomopoulos, Bourka and Themelis, 2009), there is uncertainty about whether there will be a consistent price level. Thirdly, the combustion of marine debris could lead to emissions of noxious gases. Policy makers have feared that these would have reduced health in areas proximate to WTE plants and firms have feared that these would have an associated emissions trading cost (Kothari, Tyagi and Pathak, 2010). These factors have further led to a scarcity of WTE plants. However, these fears are unfounded as evidence suggests that WTE emissions are 60% lower than using fossil fuels. Specifically, converting one tonne of municipal waste into electricity, generating approximately 600 kilowatt hours (KwH), is cleaner than generating the same amount of electricity using a barrel of oil. For relevance to this research, municipal waste is assumed to be consistent with marine debris (Lebreton et al. (2018), Psomopoulos, Bourka and Themelis (2009)). Therefore, the emissions problem, while having been a major issue against the construction of more WTE plants (DEFRA, 2014), is of low concern. Finally, an argument against the introduction of marine debris to WTE plants is the infrequent collection of marine debris from passive system The Ocean Cleanup (2014) scheme which suggests WTE plants inefficiently sitting idle. Therefore, while a clean generation of power and integration with existing infrastructure are advantages of using WTE, the low and volatile electricity price is a significant factor in suggesting that WTE not be currently a financially viable plan for the use of recovered marine debris from TOC.

Recycling.

Recycling is an attractive option for recovered marine plastics as it improves the lifespan of plastics and therefore is an efficient use of existing resources. Notably, manufacturers may prefer to use recycled plastics in their processes to reduce effluent which implies that recycling is an environmentally friendly option (Bohm et al., 2010). However, a range of factors indicates that recycling is not a financially viable use for plastics. Firstly, recycling like pyrolysis only applies to plastics which reduces the effective yield and introduces a cost in sorting debris. Indeed, WRAP (2012) reported significant costs in sorting, preparing and preprocessing debris before it is recycled which effectively reduces the profitability of recycling. The profitability of recycling is also hindered by the fact that the pricing of recycled plastics varies significantly with type and condition (Psomopoulos, Bourka and Themelis, 2009). The condition of recycled plastics is critical as the effect of degradation of marine plastics is to reduce the number of times they may be recycled and their yield from recycling (Bohm et al., 2010). However, in a sample of marine plastics, the observed level of degradation was not sufficient enough to eliminate recycling of those plastics, implying that this problem is limited (Lebreton et al., 2018). While the degraded quality of plastics was observed, they did not appear to be severe enough to reduce usability. However, for marine plastics that are not collected and remain marine debris for extended amounts of time, then there is a higher probability that such degradation effects reduce the usability and thus the financial viability of collecting such plastics.

A discussion of recycling is incomplete without addressing the interesting difference between recycling plastics for sale to manufacturers and recycling plastics into products for sale to consumers. Wholesale to manufacturers is an attractive prospect as TOC would only have to sell recovered plastics to a recycling plant and then the costs of converting, marketing and transporting the final plastics are owned by the plant. In essence, this option is far simpler and reduces the workload and uncertainty associated with the design and marketing of products (WRAP, 2012). However, if TOC wishes to earn higher revenues, then it may brave the more uncertain consumer market and create products from recycled marine debris (WRAP, 2009). This avenue requires that the products be priced at a profitable level. Specifically, it requires the public degree of support for environmentally friendly causes be sufficiently strong as to inflate the prices to a level whereby the high costs of recovering plastic from the ocean are met. There is some, albeit weakly related, supporting evidence for this in Dikgang (2012) who reported increased willingness to pay in consumers for environmentally friendly products. While the likely costs and prices for each scenario are discussed more fully in the results section; it is possible to note here that recycling is not likely to be a profitable use for recovered marine plastics unless consumers pay a considerable premium (Bohm et al., 2010).

This section has reviewed the literature on marine debris in four main sections the effects, control, types and uses of marine debris. Starting with the review of the effects of marine debris, this section revealed some ambiguity about the fatality of the effects of marine debris on marine life. This contrasted with the unambiguous damages suffered by different sectors of the marine economy, notably including fishing and tourism (Mouat, Lozano and Bateson, 2010). Given these damages, this review then examined the potential means of controlling marine debris. This section concluded by suggesting that the among the suitable designs, TOC stands out as low operating cost and high capacity

(The Ocean Cleanup, 2014) and is thus the most appropriate choice of scheme for further examination. However, this finding was conditional on the types and uses of marine debris recovered by TOC. The types of marine debris were mainly discussed with regards to the 60-80% that is believed to be plastic (Derraik, 2002). However, the significant non-plastic proportion, mainly, discarded fishing gear, was also discussed. This section raised the notion of a duality between the financial viability of recovering valuable plastics, and the economic viability of recovering debris that inflicts significant damages to marine life and economy. This duality was further evidenced by the final section which discussed the uses for recovered marine debris. Only WTE plants could make use of all types and conditions of recovered marine debris although this was associated with volatile electricity prices and a scarcity of plants. This scarcity was similar for pyrolysis plants which are very restrictive in what debris they will accept although their resultant oil is undoubtedly a valuable use for the recovered debris. Another potentially valuable use for recovered marine debris would be recycling although this applies only to plastics. Recycling suggests that marine debris may be integrated into existing waste-management infrastructure which reduces costs to TOC. However, recycling poses a challenging choice between selling recycled plastics wholesale or commercially (WRAP, 2012) although it is not clear whether either is profitable. Indeed, the financial and indeed economic viability of recovering and marketing recovered debris using TOC is uncertain and therefore the following section discusses the design of a suitable methodology to assess this.

Table 1: **Summary of literature review:**

This table summarises the impacts and uses of marine debris as described in the literature. Seminal sources include: Mouat, Lozano and Bateson (2010), Leggett et al. (2014), DEFRA (2014), Psomopoulos, Bourka and Themelis (2009), Fivga and Dimitriou (2018), Kothari, Tyagi and Pathak (2010), McIlgorm, Campbell and Rule (2011).

Impact	Description	Frequency
	Environmental	
Ingestion	Marine life may ingest floating plastics mistaken for typical diet	30% of marine life
Entanglement	Marine life may become entangled in discarded fishing gear	40% of marine life
Leaching	Leaching of toxic contaminants when plastics degrade	Unknown
Transportation	Invasive species could be transported by floating debris	Unknown
	Economic	
Lost fish stock	The above impacts all reduce fish stock	86% of fishing vessels suffered reduced catch
Reduced tourism	Washed ashore debris deters tourists	Magnitude varies by location although consistently significant
Damaged vessels	One incident per vessel per year on average	Rescue costs in the UK ranged from $1.07-2.71mn annually
Loss of non-use value	Loss of debris free areas	Value varies with area and valuation method
Reduced water quality	Leaching of chemicals and concentrations of marine debris	Not currently assessed
Costs of clean ups	Recovery of beach debris onshore Recovery of marine debris at sea by TOC	Beach cleanups: $50,710 annual average Fishing for litter: $2.24 per kg TOC: $0.97-5.03 per kg
Cost to installations	Damaged equipment at harbours and ports	€8,000 per harbour per year in UK
	Use	
Pyrolysis	Transformation of plastics to crude oil The oil may be further refined, sold or used	Cost between $0.039-1.14 per kg Prices range from $ 0.42-0.98 per kg
Waste-To-Energy	Combustion of all debris for generating electricity	Costs not stated. One tonne of debris generates $0.102 worth of electricity
Recycling	B2B: Sell to PRF B2C: Pay PRF for recycling then sell final product	B2B: $0.21 profit per kg B2C: $0.09 profit per kg

Methodology

The review of the literature suggested a dichotomy between the financial and economic viability of recovering marine debris. The following section discusses plausible methods of assessing both the financial and economic viability of TOC. To do this, a cost-benefit analysis was undertaken, and the data for this was assessed using value-transfer.

CBA.

This section evaluates the merit to using CBA for this research. To define, CBA is a technique used to assess the economic efficiency of a proposal. Economic efficiency implies that the benefits of operating a proposed scheme exceed the costs. Benefits and costs may, however, be defined in two ways, financial and economic. Financial costs and benefits refer merely to the private expenses and revenues of a proposal. Economic costs and benefits, however, refer more widely to the social costs and benefits, i.e. the impacts on society of a proposal. Specifically, economic costs and benefits include welfare issues, non-use and aesthetic values and health impacts among other items which are not included in the purely financial costs and benefits (Vatn and Bromley, 1994). This apparent difference between the two notions of economic efficiency was addressed in this research by assessing both the financial and economic viability of TOC. Assessing both was advantageous as the policy implications from each notion of viability are wide-ranging. For instance, a finding of financial viability could precipitate new private sector involvement in pollution abatement. Alternatively, a finding of economic viability would suggest a different funding source such as governmental, charity or crowdfunding. Therefore, this research assessed both financial and economic viability. Indeed there are an array of similar examples in the literature, especially in the area of environmental economics (Costanza, 2007). However, the validity of assessing both types of viability is not without controversy.

While determining the viability of a scheme by comparing all the possible costs and benefits is theoretically sensible, it is a technique not without its deficiencies. One of the most notable opponents of the method, Ackerman and Heinzerling (2002), refined their critique into the pithy summation:

"Cost-benefit analysis cannot overcome its fatal flaw: it is completely reliant on the impossible attempt to price the priceless values of life, health, nature, and the future" (Ackerman and Heinzerling, 2002, p. 1533).

This criticism has three critical elements, pricing the priceless, valuing life and welfare, and valuing the future. Firstly, pricing the priceless refers to the potentially erroneous comparison of market and non-market values. While market values are both spatially and temporally sensitive and therefore are volatile, they are also readily accessible which is analytically convenient. This convenience contrasts with non-market values which also vary strongly with time and location, but also with participant, elicitation method and object. Given this greater variability in non-market values, it is not surprising that comparison of non-market values with market ones is a notable criticism from Ackerman, Heinzerling and Massey (2005). Indeed, this issue of valuing non-market items is one that consistently surfaces in the related literature (Vatn and Bromley, 1994). This was corroborated by Ackerman and Heinzerling (2002) who noted that the various methods of eliciting non-market values do not produce estimates that are sufficiently accurate as to be included in a CBA and still be regarded as valid. This perceived inaccuracy stems from the problems of valuing items in a survey or experiment as opposed to market values, an effect referred to as hypothetical bias (Johnston et al., 2015). This critique was mitigated in this research by implementing adjusted unit value-transfer to ensure the accuracy of the final estimates, rather than relying on potentially questionable methods used in the original studies.

The second critique from Ackerman and Heinzerling (2002) was to dispute the focus on utility maximisation in CBA decision rules. Whereas maximising expected utility and using a utilitarian decision rule, the greatest good for the greatest number, is analytically convenient, it fails to incorporate any sense of equity and morality. Instead, Ackerman, Heinzerling and Massey (2005) suggested that incorporating distributional or welfare impacts would be an improvement. These impacts have high importance in environmental economics as natural resources have benefits for wider society yet may still be exploited if only private costs are considered. An example of such an erroneous finding, suggested by Ackerman and Heinzerling (2002), was the study which found that the government should promote smoking as it was a cost-effective way to reduce health expenditure. This finding failed to consider any welfare issues accompanying the value of life and health, and thus the valuations were too low leading to an erroneous judgement. Therefore, Ackerman and Heinzerling (2002) argued that CBA could not be a valid method if it failed to adequately consider impacts on society. This critique is compounded by the difficulties in estimating welfare impacts as evaluation of these non-market notions is also the basis of the very first critique Ackerman and Heinzerling (2002) made. To address this quandary, this research aimed to incorporate the impacts of marine debris wherever possible in the CBA. However, as original data could not be collected, previous estimates for each of the effects had to be relied upon to satisfy this critique which implies a degree of inaccuracy in valuing welfare impacts of marine debris in this research.

Finally, Ackerman and Heinzerling (2002) critiqued the valuing of the future via the use of Net Present Value (NPV). Net Present Value (NPV) is a technique which discounts and then aggregates all the values throughout the lifetime of a scheme to allow them to be comparable with present values. Present values cannot correctly be compared with future values for two reasons. Firstly, there is increased uncertainty about long-running schemes which requires future values to be higher to account for this uncertainty. Secondly, agents are strongly presently biased and thus weight present values more highly compared to future ones (Frederick, Loewenstein and O'donoghue, 2002). The solution to these two issues is to compare present values with NPV where NPV is calculated by discounting all the costs and benefits in the lifespan of a project (Costanza, 2007). In this context, discount rates refer to how strongly immediate values are preferred to future ones. Specifically, a higher discount rate implies that future values must be much larger than present values for agents to disregard their present bias and prefer future values. Correspondingly, lower discounting implies a greater willingness to wait for delayed values (Zhuang et al., 2007). Given the differences that magnitudes of discounting make, the correct choice of discount rate in this research is crucial. Using too low a discount rate was criticised by (Ackerman, Heinzerling and Massey, 2005, 1533) as "improperly trivialising the future". This research avoided trivialising the future with inappropriate discounting by consulting the literature for appropriate choice of discount rates. However, this was complicated by selecting different discount rates for the financial and economic effects. This research opted to use a lower discount rate for the economic impacts of 3.5% compared to the 5% for the financial impacts. The 3.5% discount rate for the economic impacts was selected Zhuang et al. (2007) explicitly suggesting that a lower discount rate should be chosen for economic impacts than financial ones. An explanation for this is that economic impacts have a longer duration than financial ones as they may accrue more slowly and thus must be discounted at a lower rate, and thus 3.5% appears to be a sensible choice (Costanza (2007), Zhuang et al. (2007)).

Moving to the discounting of the financial values, this choice of discount rate had to be both higher than that used for economic impacts and had to be justified in the literature. 5% was chosen for the financial impacts to represent bank lending rates for the representative period and sum. Indeed, many banks were observed to be charging loans at $\sim$ 4% margin over the central bank rates which would imply support for a figure in the 4-5% range. Further justification for the selection of 5% was that in their review, Zhuang et al. (2007) reported that discount rates between 2-8% had been commonly used. More specifically, Costanza (2007) corroborated this by using a 5% discount rate alongside a 3% discount rate in their discounting of marine benefits. So while the specific discount rates are justified, further criticism of using discounting is assuming a constant discount rate across time rather than accounting for diminishing preferences over time. As time periods pass, present bias weakens, so agents are more willing to wait for delayed values. Thus discount rates have often been observed to not be linear in time, such as the constant use of 3.5% for each of the ten years, but instead, either exponential or hyperbolic which would imply a diminishing discount rate for each additional year of the scheme (Frederick, Loewenstein and O'donoghue, 2002). However, as TOC is relatively short run, ten years compared to 30-50+ of some projects (Costanza, 2007), the strength of diminishing discounting is not likely to be a strong one and thus a constant discount rate is a valid assumption to make for this research. To summarise, the financial impacts of TOC were discounted at 5% annually and the economic impacts at 3.5% annually to reflect present bias and differences between economic and financial impacts.

The above section concluded that it is valid to use CBA in this research provided that appropriate measures be taken to ensure the validity of the estimates used. This section discusses how appropriate steps were taken to ensure the validity of the estimates. To assess the financial viability was relatively straightforward. Firstly, data on the anticipated expenses of the scheme are sought. These were readily available from The Ocean Cleanup (2014) although required adjustment for accuracy. Following this, it was necessary to find data on the anticipated revenues of the scheme. Yields and costs were available from the literature and prices were available from the market, and the full range of sources allowed a weighted average to be estimated thus increasing the accuracy of the estimates (Fivga and Dimitriou (2018), DEFRA (2014), Bohm et al. (2010), 2WG (2018), WRAP (2018)). Finally, the estimates were then compared in NPV terms to incorporate suitable discounting.

The above section has endorsed the use of CBA. The following section discusses the measures used to gather data for use in this CBA. As primary data was unavailable, this research followed the examples of Costanza (2007), Johnson (1994) and Brouwer and Spaninks (1999), in gathering all previous relevant estimates of the impacts of marine debris in a meta-analysis. However, this meta-analysis suffered from small sample size due to a scarcity of literature. Therefore, the purpose of the meta-analysis was to identify estimates that may be suitable for value-transfer. Indeed, Brouwer et al. (1999) suggested that combining a meta-analysis with an appropriate form of value transfer is a 'second-best' approach when compared to collecting primary data valuations. However, without primary data available, meta-analysis combined with value-transfer was chosen as a method to estimate values for use in the CBA.

Meta-analysis.

Meta-analysis collates prior estimates to synthesise new ones. Typically, this would require including a large number of studies in a meta-analysis to improve the accuracy of the final estimates by addressing shortcomings in each study. However, this research is in a novel area with a scarcity of literature, so here the best-practice use of meta-analysis was not to synthesise estimates but to indicate seminal studies for value-transfer (Costanza, 2007). Therefore, the meta-analysis in this research was limited to a small sample size which implied a high degree of reliance on fewer estimates, as opposed to taking advantage of several independent estimates as in a typical meta-analysis. To compensate for this low sample size, this research adhered to Brouwer et al. (1999) guidelines for the best practice of conducting a meta-analysis. Specifically, Brouwer et al. (1999) suggested that to ensure accuracy, meta-analysis can, and in this research should be combined with value transfer, the process of translating previous estimates into being relevant to a new scenario (Costanza, 2007). The validity and use of value-transfer is discussed next, although here it suffices to note that the purpose of meta-analysis in this research was to identify studies for suitable for value-transfer, rather than for synthesising all the estimates as typically used (Brouwer and Spaninks, 1999).

Value-transfer.

Value-transfer is a core part of this research as it allows values to be estimated for use in the CBA despite an absence of data on social and welfare impacts, unavailability of primary data, and the age of the original estimates. For reference, value-transfer in this context may be defined as:

"Transposing monetary, environmental values estimated at one site (study site) through economic valuation techniques to another (policy site)" (Brouwer and Spaninks, 1999).

Value-transfer may be undertaken using a variety of techniques although this research mainly focussed on the technique of adjusted unit value-transfer. Adjusted unit value-transfer refers to adjusting the units in a method so that the transposing of estimates from one policy site to another is reflective of differences and changes in the original and new study sites (Brouwer et al., 1999). The original policy site was the Asia-Pacific (APEC) region in 2008 where annual losses were estimated at $1.25bn using McIlgorm, Campbell and Rule (2011) method. To transpose these estimates into being relevant to the APEC region, 2018 requires adjusting McIlgorm, Campbell and Rule (2011) method to reflect the increased size and activity of the marine economy since 2008 (Hammer, Kraak and Parsons, 2012) and also to reflect the increased frequency and value of marine debris in the APEC region since 2008 (Jambeck et al., 2005). Therefore, the adjustments in this research were upwards revisions of McIlgorm, Campbell and Rule (2011) method to reflect the likelihood of higher damages. Moreover, upwards revisions of the economic impacts on the marine economy are advisable as a means to incorporating the absence of valuations regarding the social and broader economic impacts of marine debris. Therefore to mitigate their absence, an upwards adjustment of the units improves the accuracy of the estimates used in the CBA. While the specific adjustments made are detailed in the results section, the following discussion examines the validity of using adjusted unit value-transfer to estimate values for use in a CBA.

Value transfer is a convenient although contentious method. Value-transfer was hailed in Costanza (2007) as a convenient and cost-effective method of value elicitation where a primary data collection, which is typically regarded as the 'first-best' solution (Brouwer and Spaninks, 1999), is not possible. Indeed, this technique has support as several previous empirical estimates in the field of environmental valuation have combined meta-analysis and value-transfer to estimate values for use in a CBA (Brouwer et al., 1999). However, despite the popularity of the method, Brouwer et al. (1999) noted a series of criticisms. In particular, they cast doubt on the validity of estimates from value-transfer as efforts to validate generated estimates have thus far failed. This doubt may be further exacerbated where value-transfer is applied to elicited non-market values. As discussed above, non-market values are vulnerable to criticism, notably by Ackerman, Heinzerling and Massey (2005), as the requisite methods of elicitation are prone to hypothetical bias (Johnston et al., 2015) which implies that the final values to be adjusted are inaccurate even before the inaccurate process of value transfer. This potential transfer error is cause to doubt the validity of value transferred estimates.

Another critique of value-transfer regards the assertion that if the original and new sites are not sufficiently homogeneous then the applicable value-transfer method should be adjusted unit otherwise transfer errors and inaccurate estimates arise (Brouwer and Spaninks, 1999). In this research, adjusted unit value transfer was used as there are considerable differences in the APEC regions 2008 and 2018 (Jambeck et al., 2005). Adjusting previous estimates appears to be a crude manipulation of original estimates into being relevant to current estimates, and this problem is further compounded by lack of guidance in how to adjusted units. Thus, it may often appear that adjustments are arbitrarily decided. Indeed, this is why many researchers have favoured more complex value-transfer methods, such as benefit-transfer. However, this research is constrained by a severe scarcity of previous literature which necessitates reliance on McIlgorm, Campbell and Rule (2011) method for estimating the damages of marine debris and this was particularly amenable to adjusted unit value-transfer rather than more complex methods. To address the critique of arbitrary adjustments, this research endeavoured to justify each adjustment made to McIlgorm, Campbell and Rule (2011) method. Furthermore, as McIlgorm, Campbell and Rule (2011) only estimated the financial impacts of marine debris on the marine economy and not the broader social impacts, any adjustment errors in this research are still likely underestimates of the actual social, i.e. economic, impacts of marine debris. Therefore, while debatable, this research implements adjusted unit value-transfer as a practical and justifiable method of generating estimates.

As a final aside regarding the accuracy of the estimates used in the CBA, this research applied sensitivity analysis to the estimation process wherever possible. This is particularly evident in the results section where ranges of expenses, revenues, costs and benefits are produced rather than singular estimates. The purpose of this, notably in the financial analysis, was to increase the margins of error in the estimates. As this research relied on value-transfer of a single study, it is instructive to consider all possible methods of increasing accuracy in the conclusions which here implies a need for this sensitivity analysis. The effect of this sensitivity analysis was to add strength to the findings of financial infeasibility and economic viability, given that these conclusions are shown to be robust even to severe deviations.

Summary.

To summarise the method adopted in this research, a cost-benefit analysis was used to assess the financial and economic viability of TOC. To estimate values for use in these two CBA's, a meta-analysis was used to identify which studies are most amenable to value-transfer. Indeed, the use of value-transfer was a vital determinant of the validity of the estimates used in the CBA. Specifically, adjusted unit value transfer was used to compensate for the unavailability of primary data and the small sample size for the meta-analysis. Whilst the specific adjustments and their rationale is discussed later, it suffices to note that the accuracy of the method for eliciting estimates for this the two CBA's, one on the financial viability or profitability of TOC, and an expanded version which determined the economic feasibility of the scheme, was bolstered by use of adjusted unit value-transfer. This use of adjusted unit value transfer allows the valuations used in the expanded CBA to be more reflective of the full damages of marine debris in the modern APEC region. The following section discusses the results from this methodology at length.

Results

The following section calculates and analyses the financial and economic viability of TOC. Firstly, the potential profitability of the scheme is explored before the analysis is expanded to incorporate the full economic impacts of the scheme. From this, the scheme is shown to be economically but not financially viable.

Financial viability

Costs

This section examines the expenses and revenues involved with operating TOC. The expenses, predicted from The Ocean Cleanup (2014), were subject to adjusted unit value-transfer in this research. Following this, the revenues from use of recovered marine debris were also estimated using the current market prices for oil, electricity and plastics. After both expenses and revenues were calculated in per kilogram (kg) and in Net Present Value (NPV) terms, a determination of the financial viability of the scheme was possible. This section concludes by finding that the scheme cannot be profitable with the current high operating expenses and the low market prices for recovered marine plastics.

Regarding the estimated expenses, the costs reported in Tables 2 and 3 were adapted from The Ocean Cleanup (2014) with four adjustments. The first adjustment was that the original operating costs were re-estimated to be variable with capacity instead of assuming a constant operating cost. The assumption of constant operating cost was determined to not be valid in this research as the category of operating costs included fuel, maintenance and crew costs, which should theoretically all vary with the amount of plastics recovered rather than being fixed with capacity. To reflect this change in assumptions, this research adjusted the operating costs to calculate a constant operating cost per kilogram. This required using the costs from the base scenario, the $5,699,690 (Table 2), divided by the 7,000 tonnes of marine debris recovered to calculate a constant operating cost per kilogram of $0.814 which was then extrapolated to greater capacities to estimate all the operating costs, as seen in Table 4. A constant cost per kilogram was chosen to reflect linear economies of scale in recovering marine debris. While increasing returns to scale could also have been assumed, there was no indication in the initial feasibility study as to the existence or magnitude of such economies. In summary, the first adjustment was to recalculate the operating costs to be variable.

The second cost modification regarded adjusting the deviations used for sensitivity analysis. The original data adopted an 18% decrease in costs for the best case and a 16% increase for the worst case. This asymmetry was removed in this research and instead a uniform 20% deviation was estimated for both scenarios. This wider deviation allowed for a wider margin of error to be calculated along with adding power to the ability of the sensitivity analysis. This controlled for inherent uncertainty in the scheme by suggesting a range of values rather than singular estimates.

The third modification to the expenses was the use of discounting. As discussed in the methodology, discounting is an essential technique for comparison of future values with present ones using the technique of Net Present Value (NPV). Where NPV has been oft used in the literature, a common challenge has been the incorrect choice of discount rate. To avoid this critique, this research adopted different discount rates for the financial and economic effects and consulted the literature for guidance on the appropriate levels of discounting. Specifically, a 5% discount rate was used to calculate the NPV of the financial values, to reflect a greater preference for immediate revenues, whereas a 3.5% discount rate was used for the economic impacts which stretch further into the future. Indeed, discounting environmental impacts at a lower rate than financial impacts is contemporary of the literature (Zhuang et al. (2007), Costanza (2007), Frederick, Loewenstein and O'donoghue (2002)). Therefore, the financial impacts were adjusted for a 5% discount, and the economic impacts were adjusted using a lower 3.5% discount rate. However, if future researchers modify these, it is unlikely to alter the conclusions significantly as evidenced in Table 18.

The fourth and final cost adjustment used in this research was the estimation of break-even (B-E) prices. Given the expense adjustments discussed above, a total cost figure in NPV terms can be calculated. From this, dividing by the total amount of recovered marine debris allows for an expense per kg recovered to be calculated. Such a figure is instructive for a more straightforward comparison against revenues as they were typically priced in kg terms. Evidence of this was the $4.19 estimate which was calculated by dividing the NPV of the expenses, the $293mn, by the total amount of marine debris recovered, 7,000 tonnes annually. Deviations on this estimate are also reported in Table 3 for completeness. The following section now details the expense estimates before estimating a NPV revenue per kg for comparison.

Table 2: **Annual costs:**

This table reports the cost estimates from The Ocean Cleanup (2014) for a single year. While decommissioning was only incurred in the final year it is hear included to spread the cost over the entire duration. No discounting was used here as this is only a single year. However, following results update this.

Category	**Base cost**	**Best (-20%)**	**Worst (+20%)**
Capital expenditure	$20,542,008	$16,433,604	$24,650,409
Operating expenditure	$5,699,690	$4,559,752	$6,839,628
Decommissioning	$1,916,340	$1,533,072	$2,299,608
Total inc. misc costs	$36,160,611	$28,928,489	$43,392,729
Cost per kilogram [1]	$5.17	$4.13	$6.20

[1] Assuming 7,000 tonnes recovered per year

Table 3: **Total costs:**

This table reports the total cost estimates over the lifetime of TOC. Discounting at 5% was adopted here to calculate the net present value of the scheme.

Category	**Base cost**	**Best (-20%)**	**Worst (+20%)**
Capital expenditure	$205,420,080	$164,336,048	$246,504,096
Operating expenditure	$56,996,904	$45,597,524	$68,396,280
Decommissioning	$19,163,400	$15,330,720	$22,996,080
Total inc. misc costs	$361,606,112	$289,284,896	$433,927,296
NPV [1]	$293,183,808	$234,547,040	$351,820,512
Cost per kilogram [2]	$4.19	$3.35	$5.03

[1] 5% discount rate assumed here to exceed the 3.5% used for the economic impacts. A 10 year lifespan was also assumed following guidance from The Ocean Cleanup (2014).

[2] The NPV estimates were divided by the estimated 7,000 tonnes recovered annually. This capacity estimate is later adjusted for sensitivity analysis.

Table 4: **Updated costs:**

This table reports on all the amendments made to the figures from The Ocean Cleanup (2014). The ± 20% deviation is used for sensitivity in the cost estimates. Additionally, the operating cost is recalculated to be variable with capacity. Finally, B-E prices are calculated from dividing NPV by capacity. The conclusions from these scenarios are consistent.

Capacity	**Cost**	**Base cost**	**Best (-20%)**	**Worst (+20%)**
7,000 t/y	Operating	$56,996,904	$45,597,524	$68,396,280
	Totals	$361,606,112	$289,284,896	$433,927,296
	NPV	$293,183,808	$234,547,040	$351,820,512
	Cost per kilogram	$4.19	$3.35	$5.03
15,000 t/y	Operating	$122,136,224	$97,708,984	$146,563,456
	Totals	$426,745,408	$341,396,320	$512,094,528
	NPV	$345,997,568	$276,798,048	$415,197,120
	Cost per kilogram	$2.31	$1.85	$2.77
45,000 t/y	Operating	$366,408,672	$293,126,944	$439,690,400
	Totals	$671,017,856	$536,814,304	$805,221,504
	NPV	$544,049,280	$435,239,488	$652,859,264
	Cost per kilogram	$1.21	$0.97	$1.45

[1] A constant operating cost per tonne was derived by dividing the operating costs ($56,996,904), assuming base cost and 7,000 tonnes recovered, by the 7,000 tonnes for a constant operating cost per tonne of $814. This was then extrapolated for both the 15 and 45,000 tonne scenarios as well as the ± 20% deviations.

Table 5: **NPV of expenses only:**

This table reports the NPV of the estimated total expenses. The base cost 7,000 tonnes capacity scenario is used as The Ocean Cleanup (2014) believed this to be the most likely scenario. The NPV of the costs is $293.2mn although to truly be net value, the revenues should be included as well.

Year	**Annual costs**	**Discount factor**	**Present Value**
1	$36,160,611.20	0.95	$34,424,901.86
2	$36,160,611.20	0.91	$32,797,674.36
3	$36,160,611.20	0.86	$31,242,768.08
4	$36,160,611.20	0.82	$29,760,183.02
5	$36,160,611.20	0.78	$28,349,919.18
6	$36,160,611.20	0.75	$26,975,815.96
7	$36,160,611.20	0.71	$25,710,194.56
8	$36,160,611.20	0.68	$24,480,733.78
9	$36,160,611.20	0.65	$23,323,594.22
10	$36,160,611.20	0.61	$22,202,615.28
NPV [1]	$293,183,787.29		

[1] 5% discount rate used

Revenues
Crowdfunding.

This section discusses the four possible revenue sources for TOC, assuming their technology is useful. The first source of revenue is crowdfunding, and while this is not related to output, it has been used by TOC before so it is possible that crowdfunding could continue to be used. Crowdfunding has become an increasingly popular method of fundraising, which mainly operates by asking the public to donate however much they wish to a scheme. The flexible donation amounts allow for an increased number of donors who may all contribute their marginal willingness without any social pressure. Additionally, social media campaigns and charity use of crowdfunding has allowed crowdfunding platforms to host a range of sponsoring efforts for charities worldwide with great success. As such, the TOC opted to fund both the feasibility study and initial research and development for the scheme via a crowdfunding campaign. This campaign capitalised on strong public support for environmental policies which had been bolstered by previous such policies, an example of behavioural spillover effects (Dikgang, 2012). Such strong public support was evidenced by the success of the campaign that managed to raise \$2,154,282 in 100 days. As the campaign only lasted 100 days, it is however uncertain whether crowdfunding would be a sustainable source of funding for the scheme over the ten-year expected lifespan. Another challenge to calculating revenue from crowdfunding figures is the issue of how the results of the first campaign may be amended. This research uses the same $\pm$ 20% deviation as for the costs to estimate sensitivity in the crowdfunded revenue. However, as the campaign only ran for 100 days, it is perhaps erroneous to cite results from that for a full year. Future researchers may choose to calculate an average daily crowdfunded revenue amount, by dividing \$2,154,282 by 100 days, and then extrapolating for as many days as required. However, this method faces uncertainty over the duration of the campaign. If the campaign runs indefinitely or for the full ten years lifetime, there is no guarantee that the daily figure will remain constant over this extended period. Therefore, this research opted to report the crowdfunded revenue from the 100 days campaign as estimated funds raised annually for lack of other estimates.

Pyrolysis.

Of the three possible uses for recovered marine debris, pyrolysis appears to possess the highest potential for financial viability. For calculation of the potential revenues from pyrolysis, there is a range of plant size and oil types which determine the profitability (Fivga and Dimitriou, 2018). To approximately estimate the potential revenues from pyrolysis, this research used the average costs (range \$0.039-1.14 per kilogram) and the average oil prices (\$0.60) with zero refinement costs. Following this, Table 9 reports that the highest possible revenue from pyrolysis, using favourable conditions, predicts an annual revenue of \$4.7mn which is less than 15% of the annual costs of the scheme. Precisely, the NPV of using pyrolysis is \$15.34mn which represent a little more than 5% of the NPV of the estimated expenses of operating TOC. This lowly figure was under the most favourable conditions, and with more expensive plants and reduced oil prices, this proportion diminishes even further. Therefore, it is clear that pyrolysis is an insufficient source of revenue. For sufficiency, oil prices per kilogram would have to nearly double and while oil is a volatile commodity, such a favourable increase in prices is unlikely. Such an increase in oil prices could potentially imply that it is cost-effective to fuel the scheme by using pyrolysis oil in substitute for market crude oil. However, this is not a financially viable plan as the market prices are significantly lower than both the pyrolysis cost and the cost of recovering plastics.

Waste-To-Energy.

The elimination of pyrolysis as a profitable use for recovered marine debris then leads to a discussion of Waste To Energy (WTE) as an option for recovered plastics. However, similarly to pyrolysis, evidence in Table 8 suggests that the low price of electricity is not commensurate with the estimated expenses of recovering marine debris and therefore this is not a viable option. The viability was determined by assuming that one tonne of recovered marine debris yields 600-kilowatt hours (KwH) of electricity, a yield justified by empirical evidence (DEFRA (2014), Psomopoulos, Bourka and Themelis (2009), Kothari, Tyagi and Pathak (2010)). As one tonne generates 600KwH, this can be priced using the current market price of \$0.12 per kWh. Therefore, one kilogram of marine debris raises \$0.072 in revenue using WTE. This low level of revenue is insufficient to support the costs of the scheme which range from \$0.97-5.03 per kilogram and therefore use of WTE would predict severe losses, see Table 9 and Figure 1. However, this does not eliminate WTE as an option for recovered marine debris. WTE holds the potential to be state supported as a relatively clean method of power generation which is not reliant on fossil fuels and instead solves a waste management problem. While there are some concerns about the toxicity of burning recovered, and possibly degraded, plastics, the electricity from WTE has been observed to contain up to 60% fewer pollutants than that from fossil fuels which confirms WTE as a cleaner source of power. Furthermore, WTE is advantageous as there are more WTE plants than pyrolysis and therefore it is easier to integrate marine debris into the existing waste management infrastructure, saving costs when using WTE (Sudhakar et al. (2007), Psomopoulos, Bourka and Themelis (2009), DEFRA (2014)). However, the volatile but low market price for electricity strongly implies that WTE is not a financially viable option for recovered marine debris.

Recycling.

Following the elimination of pyrolysis and WTE, table 8 then examined the profitability of recycling. There are two potential revenue streams for recycling. Firstly, wholesale recycling (B2B) sells recovered plastics to a Plastic Recovering Facility (PRF) who sell it on to manufacturers (WRAP, 2012). Wholesale is an attractive option for TOC as the costs and risks associated with the marketing and sale of products are left to the PRF where TOC would only have to transport plastics and sell them to PRF's. Selling to PRF's could be done via contracting to ensure certainty or selectively to take advantage of competition between plants and volatile markets. A further advantage to this stream is that PRF is more common than WTE and pyrolysis plants which imply easier integration of marine plastics into existing waste management. To estimate the likely prices to be faced in this method, a weighted average of wholesale market prices was derived used prices from WRAP (2018) and weights from Lebreton et al. (2018) who reported the frequency of plastic types likely to be recovered by TOC in the APEC region. A weighted price is a valid approximation of the likely revenues given uncertainty about the types of plastics recovered and their market prices. This research estimated an average price of $0.21 per kg (WRAP, 2012), and the revenues at this price are insufficient to fund the scheme. This necessitated an examination of the second potential revenue stream for recycling, commercial (B2C) recycling. In this method, TOC would pay PRF's to recycle plastics and then retake responsibility for marketing them. This increases uncertainty, risk and therefore costs on TOC although the higher risk with B2C recycling is also accompanied by a higher reward as the market prices are higher than B2B recycling with a weighted average price of \$0.41 per kg. However, the typical PRF fee, for converting recovered plastics into final products, would be \$0.32 per kg leading to a positive profit of \$0.09 per kg which is lower than the \$0.21 estimate for B2B recycling (WRAP, 2009). While WRAP (2012) disputed the \$0.09 figure and reported that B2C profits could range up to \$0.54 per kg, this would still be an insufficiently low price. Even assuming that behavioural spillover effects (Dikgang, 2012) incentivise consumers to pay more for recycled products, it is unlikely that sufficiently high prices to make the scheme profitable can be consistently achieved (The Ocean Cleanup, 2014). Therefore recycling is not a financially viable option.

Table 6: **Weighted prices:**

This table reports the weighted prices used for the estimation of recycling revenue. The index values are various types of plastics. 90% of Lebreton et al. (2018) sample was PE and so it is given the same weighting. The other categories of plastics were equally distributed and so have the same weights here.

B2B		B2C	
Value	**Weights**	**Value**	**Weight**
\$205.76	0.9	\$444.41	0.9
\$37.41	0.01	\$1,290.00	0.01
\$68.37	0.01	\$1,096.50	0.01
\$38.70	0.01	\$155.45	0.01
\$474.72	0.01	\$903.00	0.01
\$470.85	0.01	\$443.76	0.01
\$290.25	0.01	\$516.00	0.01
\$354.75	0.01	\$1,225.50	0.01
\$125.13	0.01	\$1,032.00	0.01
\$251.55	0.01	\$193.50	0.01
\$457.95	0.01	\$32.25	0.01
\$78.05	0.01	305.73	\$0.01
Total:	\$209.56	\$467.23	B2C Source: WRAP (2018)
Per kilogram:	\$0.21	\$0.47	B2B Source: WRAP (2009)

Table 7: **Prices:**

This table reports the estimated costs and prices for each revenue stream. Sources are detailed in the text. Costs are stated as \$0.00 where they have not been indicated in the literature. Pyrolysis cost is assumed inclusive of any refinement of oils. Future researchers may update these.

Type	Cost	Price	Average difference
Crowdfunding	\$0.00	\$0.31	\$0.31
Pyrolysis	\$0.039-1.14	\$0.48-0.90	\$0.27
Waste-To-Energy	\$0.00	\$0.102	\$0.102
B2B Recycling	\$0.00	\$0.21	\$0.21
B2C Recycling	\$0.32	\$0.47	\$0.09

Table 8: **Revenues:**

This table extends 7 and reports the estimated revenues from each of the possible uses of recovered marine debris. Crowdfunding is an uncertain stream so should not be relied upon. Of the remaining uses, pyrolysis appears to be the most profitable although the revenues are still clearly insufficient to compensate for the high costs of the scheme.

Element	Base prices	Best prices	Worst prices
Crowdfunding ($0.31 per kilogram)	$2,154,282	$1,723,426	$2,585,138
Pyrolysis ($0.27 per kilogram)	$3,927,000	$4,712,400	$3,141,600
Waste-To-Energy ($0.102 per kilogram)	$714,000	$856,800	$571,200
B2B Recycling ($0.21 per kilogram)	$1,470,000	$1,764,000	$1,176,000
B2C Recycling ($0.09 per kilogram)	$630,000	$756,000	$504,000

Table 9: **Revenues:**

This table reports the estimated NPV of all the possible revenues from uses of recovered marine debris. The range of revenues is $5.1-17.6mn which represents on average, 3.79% of the $293mn NPV of the costs. This overwhelmingly shows that the TOC scheme is not financially viable.

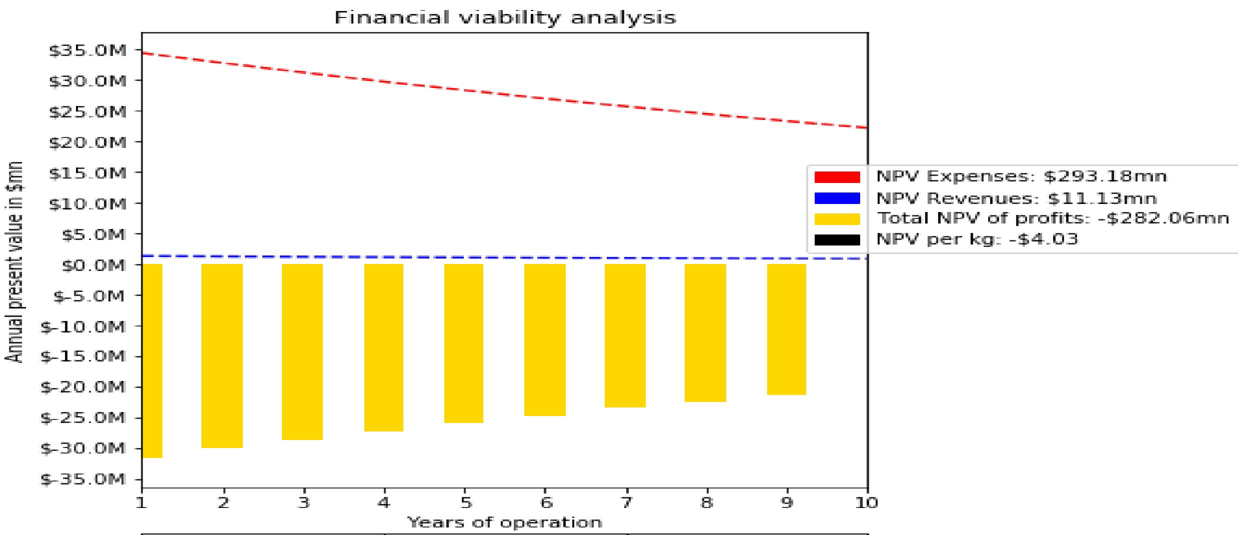
Year	**Crowdfunding**	**Pyrolysis**	**B2B Recycling**	**B2C Recycling**	**WTE**
1	$2,065,840.00	$1,801,057.07	$1,399,440.00	$599,760.00	$666,400.00
2	$1,968,190.00	$1,715,923.07	$1,333,290.00	$571,410.00	$634,900.00
3	$1,874,880.00	$1,634,572.80	$1,270,080.00	$544,320.00	$604,800.00
4	$1,785,910.00	$1,557,006.27	$1,209,810.00	$518,490.00	$576,100.00
5	$1,701,280.00	$1,483,223.47	$1,152,480.00	$493,920.00	$548,800.00
6	$1,618,820.00	$1,411,332.53	$1,096,620.00	$469,980.00	$522,200.00
7	$1,542,870.00	$1,345,117.20	$1,045,170.00	$447,930.00	$497,700.00
8	$1,469,090.00	$1,280,793.73	$995,190.00	$426,510.00	$473,900.00
9	$1,399,650.00	$1,220,254.00	$948,150.00	$406,350.00	$451,500.00
10	$1,332,380.00	$1,161,606.13	$902,580.00	$386,820.00	$429,800.00
NPV	$17,593,973.04	$15,338,917.57	$11,918,497.86	$5,107,927.66	$5,675,475.17
% total	6.00	5.23	4.07	1.74	1.94

[2] 5% discount rate used

Profitability

This section concludes by rejecting financial viability. Specifically, the tables above suggest that market prices are too low compared to operating costs which are too high. Therefore, the scheme is not profitable. However, financial viability ipso facto may be criticised as purely considering the financial impacts and not the social impacts of the scheme (Vatn and Bromley, 1994). Therefore, it is instructive to augment the finding of financial infeasibility by conducting an expanded CBA to determine if the scheme is at least economically viable.

Figure 1: **Summary of the financial viability analysis.** This graph reports the estimated expenses, revenues and the difference. The overall loss of $282mn, or -$4.03 per kg, rejects the notion of financial viability. Indeed, the ratio of revenues to expenses can be simplified to 1:24, a stunning indictment of the forecasted lossmaking nature of the TOC scheme.



	Expenses	Revenues	NPV
1	$34.42mn	$1.31mn	-$33.12mn
2	$32.80mn	$1.24mn	-$31.55mn
3	$31.24mn	$1.19mn	-$30.06mn
4	$29.76mn	$1.13mn	-$28.63mn
5	$28.35mn	$1.08mn	-$27.27mn
6	$26.98mn	$1.02mn	-$25.95mn
7	$25.71mn	$0.98mn	-$24.73mn
8	$24.48mn	$0.93mn	-$23.55mn
9	$23.32mn	$0.89mn	-$22.44mn
10	$22.20mn	$0.84mn	-$21.36mn
NPV	$293.18mn	$11.13mn	-$282.06mn

Economic viability

This section assesses the economic viability of TOC. Economic viability includes the financial and the social impacts of marine debris. While the financial impacts were valued above, this section aims to value the social impacts. There is a range of social impacts of marine debris, such as loss of life, non-use and aesthetic values (Lebreton et al., 2018), but the absence of valuations of these impacts necessitates the assumption that the primary economic impact of marine debris is the damage to the marine economy (McIlgorm, Campbell and Rule, 2008). To estimate this impact, a meta-analysis was first conducted to identify which prior studies are suitable for value-transfer. Adjusted unit value-transfer was then applied to the seminal papers from the meta-analysis to estimate the economic impacts for use in the CBA to assess economic viability.

Table 10: **Meta-analysis:**

Sources and valuations used in the meta-analysis. Meta-analysis was used to identify the unique estimates of marine debris in the literature. The seven below are the seminal papers and of these, McIlgorm, Campbell and Rule (2011) and Mouat, Lozano and Bateson (2010) are identified as viable for adjusted unit value-transfer.

Study	Methodology	Valuation	Additional calculations
McIlgorm, Campbell and Rule (2011)	Designed a novel function detailed in text	Annual losses of $1.26bn APEC marine economy	By sector: Fishing: $364mn damages Shipping: $279mn damages Tourism: $622mn damages
Kildow and McIlgorm (2010)	Similar to McIlgorm, Campbell and Rule (2011) but on a country, not regional, level	APEC marine economy value: $40.44bn Represents on average 2.51% of national GDP	Average value of marine economy: 2.5% GDP Not 3% as above
Mouat, Lozano and Bateson (2010)	Collected original data.	Cost of clean-up: €18mn annually for UK Cost of clean-up: 37% rise over 10 years Cost to fishing: 5% of total revenue Cost to harbours: €2.4mn py Cost of rescue: €830k-2.2mn annually for UK	These calculations are based on questionnaire Estimates from North Sea rather than Pacific
Takehama (1990)	Used public data	Damage to vessels: 0.3% of revenue Alternatively: ¥6.6bn in 1985 prices	No indication if the 0.3% rule is up to date
The Ocean Cleanup (2014)	Used meta-analysis	Costs of marine plastics: $13bn annually Refers to world rather than APEC region	Costs are likely under-reported due to technical challenges restricting data
Leggett et al. (2014)	Case study of Orange county	A 25% reduction would save $32mn yearly A 75% reduction leads to benefits of $52mn yearly A 100% reduction would save $148mn yearly	50% of beach debris re-appeared within 3 months Did not specify the clean-up costs.
Bergmann, Gutow and Klages (2015)	Used meta-analysis	Foregone tourism valued at €23-29mn	They note a range of impacts of marine debris but did not quantify these

While McIlgorm, Campbell and Rule (2011) function is the most notable estimate from the meta-analysis, it is also instructive to note why this research mentioned but did not adapt the findings from Mouat, Lozano and Bateson (2010). Noteworthy findings included that 82% of fishing vessels had reported marine debris contaminated catch, that 70% of harbours reported damages from marine debris, that marine debris damages were 5% of total annual fishing fleet revenue and that beach clean up costs had risen 37% in ten years to more than $50,000 per area. However, as it is not practical to estimate the number of harbours, fishing vessels or beaches at threat from marine debris in the APEC region, these figures are less relevant to this research. Furthermore, the data used to assess the damages from the marine debris was based solely on respondents estimates which suffered from low response rate, an average of one-third of surveys was returned. This low response rate implies a sample size issue that casts doubt on their estimates. Their estimates are more thorough than McIlgorm, Campbell and Rule (2011), but less easily adjusted and replicated and therefore this research chose to adapt McIlgorm, Campbell and Rule (2011) method over Mouat, Lozano and Bateson (2010).

McIlgorm, Campbell and Rule (2011) established a novel function to calculate the costs of marine debris in the APEC region. Their function included three fundamental units.

Summary of McIlgorm, Campbell and Rule (2011) function:

Step one: Take 3% of GDP as the value of the marine economy

Step two: Take 48% of this figure as vulnerable to marine debris

Step three: Take 0.3% of this figure as annual losses.

The 3% value and the 0.3% estimate are the two units that were adjusted in this research to increase the accuracy of this function.

Firstly, McIlgorm, Campbell and Rule (2011) assumed the marine economy represented on average 3% of national GDP. Secondly, 48% of the above figure was taken as the vulnerable marine economy. Finally, 0.3% of the 48% is estimated as annual losses to marine debris. However, before adopting this function in this research, several critiques of the function must be made. The first criticism was that Kildow and McIlgorm (2010) doubted the 3% proportion of GDP and instead reported that an average of 2.5% of GDP is a more likely value of the marine economy. Furthermore, some countries specifically report the value of their marine economy which makes use of a percentage irrelevant for a minority of countries. Practically, this research reported the 3% value for comparison with the original figures, but a band of 2.5-5% was also estimated for sensitivity analysis. This band of estimates was justified by Kildow and McIlgorm (2010) who reported 2.5% as the average percentage for the APEC region. However, they also noted that individual countries, China, in particular, had significantly larger marine economies. Therefore, this research opted to use 3% as suggested but also adjusted the units to include 2.5% and 5% as lower and upper bounds respectively for sensitivity analysis of the values.

Secondly, where the 48% estimate of the vulnerable marine economy stems from is unclear, however. Therefore the 48% proportion of the vulnerable marine debris was not adjusted in this research as inadequate literature exists on how to adjust it. This example of unadjusted unit value transfer introduces additional uncertainty although this mostly unavoidable for this specific unit. The adjustment of the remaining two units should however somewhat mitigate the transfer error implied by using this proportion unadjusted (Brouwer and Spaninks, 1999).

Third and finally, the 0.3% loss value is a point of significant contention in the literature. The source of the estimate was Takehama (1990) who observed damages from marine debris suffered by the Japanese fishing fleet in the 1980's. Two critiques of this arise. Firstly, applying a localised estimate for Japan to the entire APEC region is an example of the unadjusted unit value transfer Brouwer et al. (1999) criticised as being inaccurate. Secondly, the unadjusted figure also fails to account for the growth in marine debris since that period which has been unambiguously accompanied by inflated damages to the marine economy (Jambeck et al., 2005). Therefore, it is clear that using the 0.3% figure without further qualification or amendment is erroneous. To solve this potential error, this research used a range of 0.2-0.5%, which is skewed towards higher estimates. The range of 0.2-0.5% was skewed towards the higher values given the increased prevalence of marine debris and growth of the marine economy (Jambeck et al., 2005). A further justification for adjusting for higher units is that as estimates from McIlgorm, Campbell and Rule (2011) method are likely an underestimate of the true economic costs, an upwards revision would somewhat mitigate the problem of not incorporating the full social impacts of reduced water quality, lost marine life and negative welfare impacts on coastal populations which are not currently valued. Therefore using a higher bound of 0.5% is an attempt to include these impacts. To conclude, McIlgorm, Campbell and Rule (2011) proposed a novel function to calculate the damages of marine debris but it requires adjusted unit value transfer to be relevant to this CBA.

This section discusses some necessary critiques of the above method. Despite the convenience of McIlgorm, Campbell and Rule (2011) method, it is instructive to note that while their estimates purport to represent the entire marine economy. It is, however, doubtful that this an accurate estimate for each sector afflicted by marine debris. This is mainly because the estimated damages refer to the entire APEC marine economy and not one specific sector and so the estimates are imprecise. Furthermore, the estimates specifically relate to the financial losses suffered by the entire marine economy and therefore fails to consider the social impacts McIlgorm, Campbell and Rule (2008). The social impacts include the value of lost and damaged marine life or the loss of non-use and aesthetic value (Costanza, 2007). Nor is the value of reduced water quality or the disproportionate welfare impacts on the populations who live closest to polluted areas incorporated in their estimates. Therefore, the estimates from McIlgorm, Campbell and Rule (2011) method are likely an underestimation of the full economic impacts of marine debris. As valuations of these are not currently available from the literature, the estimates from McIlgorm, Campbell and Rule (2011) must suffice. Even by adjusting McIlgorm, Campbell and Rule (2011) upwards to reflect the omitted social impacts, the resulting estimate is likely an underestimate of the true economic costs although it must suffice here. However, even with the use of underestimates, the economic viability of the scheme is assured, and realisation of the true inflated costs would only add credence to this conclusion.

This section discusses the validity of the estimated \$465.49mn NPV of the TOC scheme. Firstly, the use of NPV required discounting the economic impacts at 3.5%, compared to the 5% discounting of the financial impacts. While the precise discount rates used can be critiqued as inaccurate or unjustified, the effect of discounting is essentially one of scaling, and so the conclusions are not significantly different regardless of choice of discount rate, see Table 19 for confirmation of this. Another possible critique of this method is the reliance upon the estimated avoided damages which were used to represent the full economic impacts of marine debris. Precisely, the avoided damages were calculated by taking the total annual value of marine losses, \$1.95bn (range: \$1.08-5.42bn), and dividing this by amount of marine debris (140,000 tonnes (Lebreton et al., 2018)), for a damage per kilogram estimate, \$13.95 (range: \$7.75-38.75). This damage per kilogram estimate was then applied to the 7,000 tonnes assumed recovered annually. Using this method reported avoided damages of \$97mn annually (range: \$54.25mn-\$271.25mn). The headline figures for this research include the point estimate of the \$1.95bn estimate of annual losses. This estimate was a simple replication of McIlgorm, Campbell and Rule (2011) method. This example of unadjusted unit value transfer has been critiqued as being wildly inaccurate by Brouwer and Spaninks (1999) so the validity of using it to estimate economic viability in this research is weak. However, this figure is merely estimated for consistency with the previous literature. Furthermore, a range of estimates using the adjusted unit value transfer and sensitivity analysis methods were calculated, and they uniformly report the same conclusion, that TOC is economically viable.

Table 11: **Damage estimates:**

This table reports the results from adjusting the units in McIlgorm, Campbell and Rule (2011) method. This leads to the estimated annual losses to the APEC marine economy from marine debris, of \$1.95bn. This represents a 54.15% increase on the original damage estimates in fewer than 10 years. This is consistent with the increase in plastic production worldwide (Jambeck et al., 2005). Data for selected countries is also shown to highlight the widespread damages of marine debris.

Element	**Value**	**Notes**
Total APEC region	Total GDP = \$45,204.85bn Marine economy = \$1,356.14bn Vulnerable value = \$650.95bn Losses = \$1.953bn	3% of GDP assumed to be marine economy 0.3% of vulnerable marine economy is estimated as losses as in Takehama (1990)
USA	Marine economy = \$320bn Vulnerable value = \$153.6bn Losses = \$460.8mn Lost fishing = \$258.5mn	Uses 1.8% of GDP instead of 3% Uses 5% of total fishing revenue as estimate for losses Follows Mouat, Lozano and Bateson (2010) method
Australia	Marine economy = \$74.4bn Vulnerable value = \$35.712bn Losses = \$107.1mn Lost fishing = \$82mn	Uses 4.8% of GDP instead of 3% Marine economy and fishing value sourced from APEC (2018)
China	Marine economy = \$802.6bn Vulnerable value = \$385.25bn Losses = \$1.16bn	Data on fishing not available Marine economy value interpreted from GDP
Japan	Marine economy = \$131.49bn Vulnerable value = \$63.12bn Losses = \$189mn	Data on marine economy interpreted from GDP Data on fishing not available

Table 12: **Omitted estimates:**

This table reports some of the valuations not incorporated in McIlgorm, Campbell and Rule (2011) method. While adjusted unit value transfer can be used to approximate the omitted valuations, it is still worthwhile reporting these areas which are in need of further study and valuations. The absence of these estimates implies that the $1.95bn figure is likely an underestimate of the true costs of marine debris. This lends further weight to the conclusion that TOC is economically but not financially viable.

Category	**Estimates**	**Proposal**
Human health	The impact of reduced water quality from leached chemicals has not been assessed	Researchers must first quantify the effects and extent of reduced water quality
Marine life	40% of species suffering entanglement 40% suffering from ingestion	CV techniques could estimate agents Willingness-To-Pay valuations to keep the current marine life and area consistent
Non-use value	This refers specifically to loss of aesthetic value in areas of highly concentrated marine debris	

Table 13: **NPV estimates of the avoided damages:**

This table reports the NPV of the estimated damages avoided by operating the TOC scheme. A 3.5% discount rate is assumed, consistent with the Treasury (2018). The resultant figure of $840.5m, combined with the NPV revenues, is almost thrice the NPV of the costs which implies economic viability of the scheme.

Year	**Annual damages**	**Discount factor**	**Present Value**
1	$97,650,000	0.97	$94,329,900
2	$97,650,000	0.93	$91,205,100
3	$97,650,000	0.90	$88,080,300
4	$97,650,000	0.87	$85,053,150
5	$97,650,000	0.84	$82,221,300
6	$97,650,000	0.81	$79,487,100
7	$97,650,000	0.79	$76,752,900
8	$97,650,000	0.76	$74,116,350
9	$97,650,000	0.73	$71,675,100
10	$97,650,000	0.71	$69,233,850
NPV [1]	$840,540,587.59		

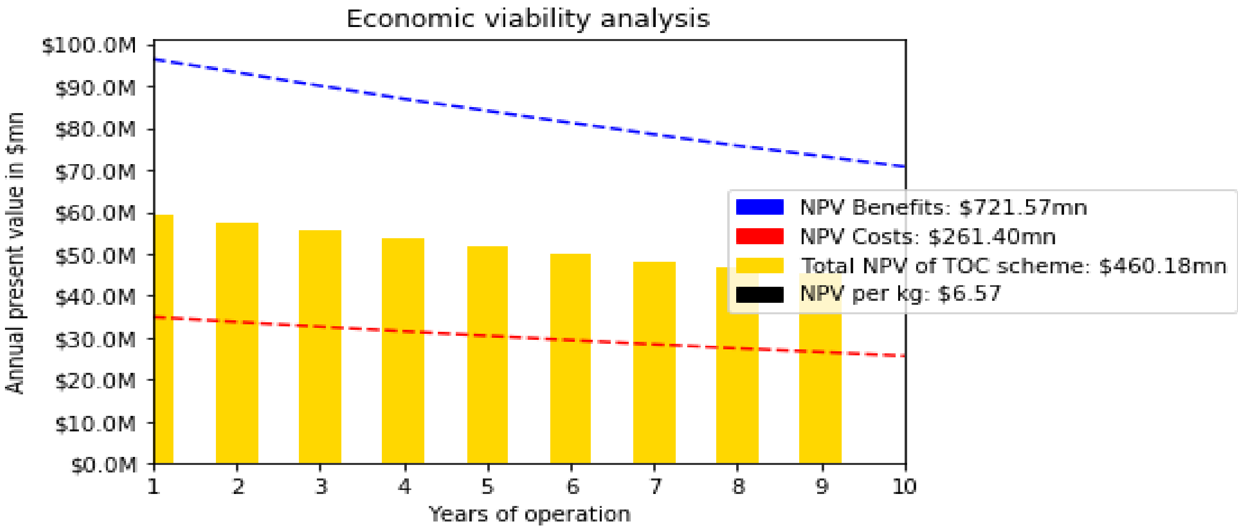
[1] 3.5% discount used instead of 5% for the financial costs.

Table 14: **Avoided damages:**

This table reports the possible adjustments to McIlgorm, Campbell and Rule (2011) method. 9 final estimates are produced for the annual value of damages to the APEC marine economy. These range from $1.08-5.42bn or $7.75-38.75 per kilogram of marine debris. These estimates of debris are later used to calculate the value of damages avoided by operating the TOC scheme.

Units:	**Estimates:**
Marine economy (M.E) value	
Total GDP:	$45,205bn (APEC, 2018)
3% (Base)	$1,356.1455bn
2.5% (Low)	$1,130.1212bn
5% (High)	$2,260.2425bn
Value of marine economy vulnerable to marine debris	
48%	$650.95bn (Base)
	$542.46bn (Low)
	$1,084.92bn (High)
Estimated annual losses	
0.3% (Base)	$1.9528bn (Base)
	$1.6274bn (Low)
	$3.2547bn (High)
0.2% (Low)	$1.3019bn (Base)
	$1.0849bn (Low)
	$2.1698bn (High)
0.5% (High)	$3.2547 (Base)
	$2.7123 (Low)
	$5.4246 (High)
Low to high ranking of estimates in per kilogram terms	
$1.08bn	$7.75
$1.30bn	$9.30
$1.62bn	$11.62
$1.95bn	$13.95
$2.16bn	$15.50
$2.71bn	$19.37
$3.25bn	$23.25
$3.25bn	$23.25
$5.42bn	$38.75

Figure 2: **Summary of the economic viability analysis.** This compares the total costs and benefits for the TOC scheme. The total costs included the financial expenses of the scheme with the main difference being that in the economic analysis the costs were discounted in the year they are incurred whereas in the financial analysis they were discounted over the lifespan of the scheme. The total benefits of TOC included the financial revenues and then avoided damages from operating the scheme. Following this, the comparison of costs and benefits showed a positive total net benefit to operating the scheme which implies economic viability.



	Total Benefits	Total Costs	NPV
1	$96.43mn	$34.93mn	$61.49mn
2	$93.23mn	$33.77mn	$59.46mn
3	$90.04mn	$32.62mn	$57.42mn
4	$86.94mn	$31.50mn	$55.45mn
5	$84.05mn	$30.45mn	$53.60mn
6	$81.25mn	$29.43mn	$51.82mn
7	$78.46mn	$28.42mn	$50.04mn
8	$75.76mn	$27.45mn	$48.32mn
9	$73.27mn	$26.54mn	$46.73mn
10	$70.77mn	$25.64mn	$45.13mn
NPV	$721.57mn	$261.40mn	$460.18mn

Summary

This concludes the results section. The tables above overwhelmingly report that while TOC is not financially viable, it should, however, proceed as it is economically viable. Financially, the high operating costs far exceed the revenues. Economically, however, the avoided damages from marine debris far exceed the losses from operating the scheme. Furthermore, as the estimated damages are likely an underestimate of the true economic impacts, the economic viability of the scheme is beyond question.

Table 15: **Extended Cost-Benefit Analysis:** Summary of the extended CBA.

Impact	Value	Range
Costs		
Lost fish stock	\$340.5mn	No range estimated
Costs of clean ups	\$293.18mn	\$234.54-652.86mn
Total annual damages:	\$1.95bn	\$1.08-5.42bn
Benefits		
Crowdfunding	\$17.59mn	\$1.72-2.58mn annually
Pyrolysis	\$15.34mn	\$3.41-4.71mn annually
Waste-To-Energy	\$5.67mn	\$0.57-0.85mn annually
Recycling	\$5.1-11.9mn	\$0.50-1.76mn annually
NPV economic benefits:	\$721.57bn	\$407.85mn-1.97bn
Summary		
NPV of entire scheme:	\$460.17bn	\$146.45mn-1.71bn
NPV per kg:	\$6.57	\$2.09-24.50

Discussion

Implications.

The implications of this research stem from the duality between the financial and economic viability of TOC. While the scheme was not remotely financially viable, the sheer magnitude of damages avoided by operating the scheme implies a degree of economic viability in the scheme. This finding implies that it is advisable to seek a sufficient source of funding to mitigate the damages of marine debris. Thus far only crowdfunding has been used. While it has been a popular source with nearly 40,000 individual donors, the revenues were less than 10% of the total annual costs [1]. Alternatively, the scheme could operate in a way where donors could sponsor the cleanup for a specific price per kilogram collected, thus directly linking revenue to operations. While this has not been suggested for TOC, this method has been successful for charities. Finally, either government or third-sector intervention is likely necessary to support this scheme. One possible solution would be the marine economy funding the scheme to mitigate the damages they suffer. Alternatively, the major industrial, commercial and maritime polluters could also support the scheme if the 'polluter pays principle' is adhered to. The precise funding mechanism is left for future researchers. A final noteworthy implication of this research is the finding that marine debris could relatively easily be integrated into the existing waste management infrastructure. Regardless of how marine debris, in particular plastics, are recovered, integration with existing waste management infrastructure should be sought as a cost-effective method of management. Recycling in particular merits further study as it represents an efficient use of existing resources and where recycled plastics substitute for virgin plastics, recycling would save fossil fuels and industrial effluent from the manufacture of virgin plastics. Therefore, future researchers are advised to analyse recycling in particular as a cost-effective method of integrating of wider waste sources, such as marine debris, into existing waste management infrastructure.

[1] The scheme did only run for 100 days to fund the completion of the feasibility study so it is not entirely accurate to suggest that crowdfunding cannot deliver revenue over 10% of the annual costs. However, the 10yr lifespan and sizeable anticipated losses suggest that crowdfunding cannot be solely relied upon to support this scheme.

Limitations.

There are, however, a range of limitations to this study. Firstly, the meta-analysis suffered from small sample size, due to lack of estimates of the social impacts of marine debris, and this small sample size leads to a strong reliance on adjusting McIlgorm, Campbell and Rule (2011) method. Furthermore, the adjustment of units in this research could be described as arbitrary. While using a band of estimates and adjusting the units upwards was justified, future researchers could choose precise adjustments where this research instead estimated ranges. A final noteworthy limitation is the strong reliance on The Ocean Cleanup (2014) despite it being a feasibility study estimated before the actual expenses of TOC are known. This reduces the accuracy of the financial viability analysis although the conclusions are likely consistent unless the TOC scheme radically changes.

Recommendations.

There are two key recommendations from this research. Firstly, the topic of marine debris requires further attention. Specifically, the scarce literature could be augmented by new willingness to pay estimates using contingent valuation techniques to evaluate the social impacts of marine debris. Secondly, the effects of leached chemicals from plastics on water quality, marine and human health require further research.

Conclusion

In conclusion, this research has evaluated the financial and economic viability of a novel scheme to recover marine debris. The TOC scheme proposed recovering marine plastics and then marketing those for profit. However, none of the revenue streams was forecast to provide even 10% of the expenses of running the scheme, and thus the finding of profitability is rejected by this research. Evidence for this is provided by the ratio of revenues to costs of 1:24 or in NPV terms, an estimated loss of $280.55mn over the lifetime of the scheme, or -$4.01 per kilogram recovered. However, while financial viability was rejected, economic viability was not. The ratio of economic benefits to costs was 1:2.78, or in NPV terms, a total net benefit of $465.49mn, or $6.65 for every kilogram of marine debris recovered. These conclusions were reached in two separate ways. For the assessment of the financial viability, The Ocean Cleanup (2014) provided estimates of the capital and operating costs.

This research amended these in four ways. Firstly, the operating costs were calculated to be variable with capacity rather than fixed. Secondly, the margins of error used for sensitivity analysis were widened. Thirdly, discounting was used. A lower rate, 3.5%, was used for the economic impacts while 5% was used for the financial impacts. Use of discounting was undertaken to compare values throughout time accurately and to allow for calculation of net present values. Finally, break-even prices, or costs per kilogram, were calculated for ease of comparison with the revenues. Of the revenues, average market prices were used wherever possible. Finally, this allowed a comparison of expenses and revenues which overwhelmingly reported that the scheme is not financially viable. For the evaluation of economic viability, previous estimates of the economic impacts of marine debris were sought. When a scarcity of estimates presented itself, this research opted to apply adjusted value transfer to a seminal paper's methodology to allow new estimates. As with the financial viability assessment, discounting and sensitivity analysis were applied for accuracy and validity. All told, the resultant estimates of the NPV of the TOC scheme are significantly positive which therefore implies the scheme should proceed. The estimates used to calculate economic benefits, however, may be critiqued as being underestimates of the true social impacts which could not be adequately estimated here for data limitations. Regardless, the presence of an underestimated economic benefit only adds weight to the conclusions of this research. With regards to the conclusion that the scheme is economically but not financially viable, future researchers must contend with a variety of solutions. Crowdfunding support has also been suggested although this is also likely insufficient to cover the schemes significant expenses. Government intervention and application of Coase theorem may also be worthwhile avenues for future researchers to explore. For now, it suffices to note that if you go fishing for litter, you better hope the price is right.

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Appendix:

Ethics.

Ethical approval for this study required disclosing of the sources used for all the data used in this research, which is also available in the bibliography and where indicated in text. Dr Eleonora Fichera, the Department Research Ethics Officer approved the ethics request for this study which was confirmed by Dr Alistair Hunt, supervisor.

Replicability.

All the data used in this research are freely available from the sources in the bibliography. Every effort has been taken to ensure calculations are explained adequately throughout so replication of results should not be a challenge. Differences may occur however if different figures are used for GDP, marine economy, fishing revenue, oil prices, recycled plastic prices, or different costs are assumed for the TOC scheme. Typically, upper and lower bounds have been estimated to mitigate this issue.

Variations on estimates in text:

Table 16: **Economic viability of pyrolysis.** An estimation of pyrolysis costs and benefits.

Pyrolysis plant data interpreted from Fivga and Dimitriou (2018). Cost of oil correct at 07/2018. Recovery cost assumed to be $5.28 (for 7000t/y with base costs).

Plant capacity (kg/h)	100	1000	10,000	100,000
Pyrolysis cost ($/kg)	1.14	0.34	0.066	0.039
Total cost inc. recovery ($/kg)	6.43	5.63	5.36	5.33
Profit to sale as: residual fuel oil ($0.38/kg)	-6.05	-5.25	-4.98	-4.95
Profit to sale as: heavy fuel oil ($0.46/kg)	-5.97	-5.17	-4.90	-4.87
Profit to sale as: marine oil ($0.73/kg)	-5.70	-4.90	-4.63	-4.60

Table 17: **Valuations:**

This table reports the complete range of per kilogram estimates. This is done for ease of reference and completeness in the results. This clearly supports the conclusions from this research.

Scenario	Price per kg	Scenario	Price per kg
Costs		**Benefits**	
Annual base cost	$5.17	Crowdfunding	$0.31
Annual best cost	$4.13	Pyrolysis	$0.27
Annual worst cost	$6.20	Waste-To-Energy	$0.102
NPV base 7,000	$4.19	Average Recycling	$0.15
NPV best 7,000	$3.35	Damages low	$7.75
NPV worst 7,000	$5.03	Damages low	$9.30
NPV base 15,000	$2.31	Damages low	$11.62
NPV best 15,000	$1.85	Damages medium	$13.95
NPV worst 15,000	$2.77	Damages medium	$15.50
NPV base 45,000	$1.21	Damages medium	$19.37
NPV best 45,000	$0.97	Damages high	$23.25
NPV worst 45,000	$1.45	Damages high	$38.75
Average cost per kg	$3.22	Average benefit per kg	$11.69

Table 18: **Summary of sensitivity analysis:** This table reports the range of financial estimates under each capacity estimate and cost scenario. The finding of no financial viability is uniformly found.

Year	Expenses	Revenues	NPV	Loss per kg
7,000 Base	$293,183,787.29	$11,126,958.26	-$282,056,829.03	-$4.03
7,000 Best	$234,547,035.02	$11,126,958.26	-$223,420,076.76	-$3.19
7,000 Worst	$351,820,513.62	$11,126,958.26	-$340,693,555.36	-$4.87
15,000 Base	$345,997,566.90	$23,843,481.98	-$322,154,084.91	-$2.15
15,000 Best	$276,798,048.33	$23,843,481.98	-$252,954,566.34	-$1.69
15,000 Worst	$415,197,111.41	$23,843,481.98	-$391,353,629.43	-$2.61
45,000 Base	$544,049,311.76	$71,530,445.95	-$472,518,865.81	-$1.05
45,000 Best	$435,239,464.98	$71,530,445.95	-$363,709,019.03	-$0.81
45,000 Worst	$652,859,236.38	$71,530,445.95	-$581,328,790.43	-$1.29

Figure 3: **Summary of sensitivity analysis** This graphs Table 18 to simply show that the scheme may potentially be financially viable if they recover 45,000 tonnes annually and have favourable cost estimates. However, Table 18 shows that revenue would also have to raise significantly.

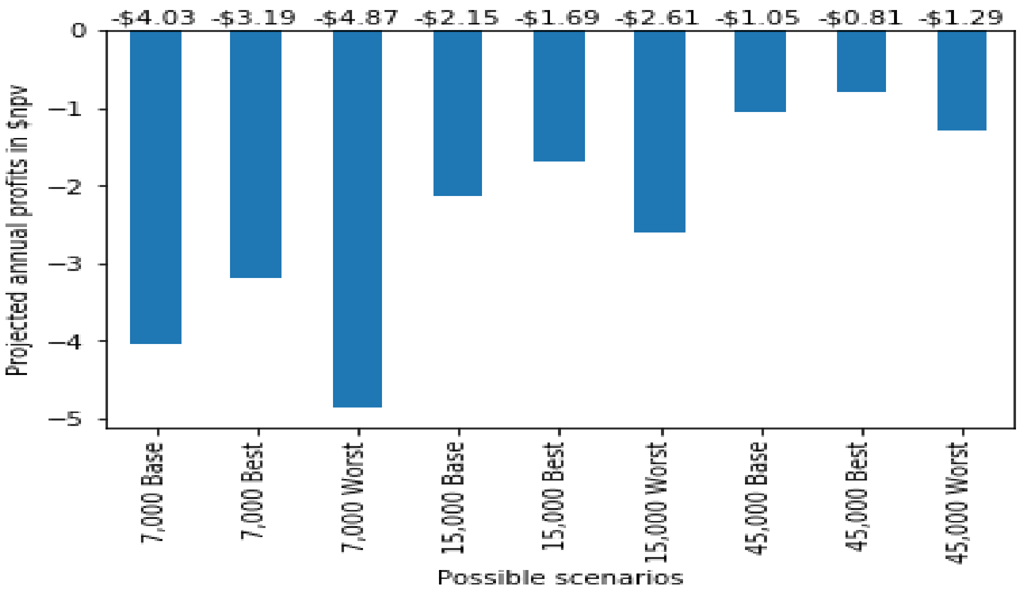

Table 19: **Summary of discounting effects:**

This table reports a range of discount rates applied to the NPV of both the financial and economic totals of the TOC scheme. From this it can be seen that discounting makes no major difference to the conclusions of economic but not financial viability.

Year	1%	2%	3%	4%	5%
Financial impacts					
1	-$34,192,656.36	-$33,505,604.75	-$32,854,713.75	-$32,203,822.75	-$31,552,931.75
2	-$34,192,656.36	-$33,505,604.75	-$32,854,713.75	-$32,203,822.75	-$31,552,931.75
3	-$33,926,222.92	-$32,877,565.19	-$31,901,228.69	-$30,961,052.80	-$30,057,037.52
4	-$33,620,884.11	-$32,282,941.50	-$30,981,159.49	-$29,787,859.32	-$28,630,719.76
5	-$33,312,800.56	-$31,685,573.05	-$-30,130,666.77	-$28,648,081.71	-$27,273,978.49
6	-$33,039,505.24	-$31,086,832.24	-$29,242,641.07	-$27,543,092.34	-$25,952,025.45
7	-$32,762,092.81	-$30,520,134.92	-$28,422,819.47	-$26,506,307.07	-$24,734,437.12
8	-$32,447,147.39	-$29,915,904.61	-$27,601,625.49	-$25,504,310.04	-$23,551,637.04
9	-$32,165,617.84	-$29,381,250.77	-$26,813,847.38	-$24,535,728.87	-$22,438,413.42
10	-$31,882,715.91	-$28,809,063.96	-$26,060,857.51	-$23,601,935.94	-$21,359,978.05
NPV	-$334,786,084.04	-$320,185,121.00	-$306,584,110.43	-$293,899,788.45	-$282,056,829.03
Economic impacts					
1	$63,022,794.91	$62,386,201.02	$61,813,266.52	$61,240,332.03	$60,603,738.14
2	$62,386,201.02	$61,176,672.64	$60,030,803.64	$58,884,934.64	$57,739,065.64
3	$61,813,266.52	$59,967,144.25	$58,248,340.75	$56,593,196.64	$55,001,711.92
4	$61,176,672.64	$58,821,275.25	$56,529,537.25	$54,428,777.42	$52,391,676.98
5	$60,540,078.75	$57,675,406.25	$54,938,052.53	$52,328,017.59	$49,908,960.82
6	$59,967,144.25	$56,529,537.25	$53,282,908.43	$50,290,917.15	$47,489,904.04
7	$59,394,209.75	$55,447,327.64	$51,755,083.09	$48,381,135.49	$45,261,825.44
8	$58,757,615.86	$54,301,458.65	$50,227,257.76	$46,535,013.21	$43,097,406.22
10	$57,611,746.86	$52,200,698.82	$47,362,585.27	$43,033,746.83	$39,086,864.72
NPV	$577,164,206.37	$525,578,169.86	$480,483,772.40	$441,171,004.28	$406,399,772.68